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Air Quality Criteria for Particulate Matter and Sulfur Oxides. Volume 3

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16. ABSTRACT

The document evaluates and assesses scientific information on the health and welfare effects associated with exposure to various concentrations of sulfur oxides and particulate matter in ambient air. The literature through 1980-81 has been reviewed thoroughly for information relevant to air quality criteria, although the document is not intended as a complete and detailed review of all literature pertaining to sulfur oxides and particulate matter. An attempt has been made to identify the major discrepancies in our current knowledge and understanding of the effects of these pollutants.

Although this document is principally concerned with the health and welfare effects of sulfur oxides and particulate matter, other scientific data are presented and evaluated in order to provide a better understanding of these pollutants in the environment. To this end, the document includes chapters that discuss the chemistry and physics of the pollutants; analytical techniques; sources; and types of emissions; environmental concentrations and exposure levels; atmospheric chemistry and dispersion modeling; acidic deposition; effects on vegetation; effects on visibility, climate, and materials; and respiratory, physiological, toxicological, clinical and epidemiological aspects of human exposure.

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Preface

This document is Volume III of a three-volume revision of Air Quality Criteria for Particulate Matter and Air Quality Criteria for Sulfur Oxides, first published in 1969 and 1970, respectively. By law, air quality criteria documents are the basis for establishment of the National Ambient Air Quality Standards (NAAQS). The Air Quality Criteria document of which this volume is a part has been prepared in response to specific requirements of Section 108 of the Clean Air Act, as amended in 1977. The Clean Air Act requires that the Administrator periodically review, and as appropriate, update and reissue criteria for NAAQS.

As the legally prescribed basis for deciding on National Ambient Air Quality Standards, the present document, Air Quality Criteria for Particulate Matter and Sulfur Oxides, focuses on characterization of health and welfare effects associated with exposure to particulate matter and sulfur oxides and pollutant concentrations which cause such effects. The major health and welfare effects of particulate matter and sulfur oxides are discussed in Chapters 8 through 14 in this volume (Volume III) of the document. To assist the reader in putting the effects into perspective with the real-world environment, Chapters 2 through 7 in another volume (Volume II) of the document have been prepared and discuss: physical and chemical properties; air monitoring and analytical measurement techniques; sources and emissions; transport, transformation, and fate; and observed ambient concentrations of the pollutants.,

Volume I introduces the criteria document, explains the rationale behind combining the evaluation of criteria for particulate matter and sulfur oxides in a single document and briefly summarizes the content of the entire criteria document. However, for a fuller understanding of the health and welfare effects of particulate matter and sulfur oxides, both Volumes II and III of the document should be consulted.

The Agency is pleased to acknowledge the efforts of all persons and groups who have contributed to the preparation of this document. In the last analysis, however, the Environmental Protection Agency accepts full responsibility for its content.

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8. EFFECTS ON VEGETATION

8.1 GENERAL INTRODUCTION AND APPROACH

The objective of this chapter is to review available data relating atmospheric concentrations of SO_2 and particulate matter (PM) to effects on terrestrial vegetation. Many reviews of the general effects of SO_2 (Daines, 1968; Guderian, 1977; Jacobson and Hill, 1970; Linzon, 1978; Mudd, 1975; National Academy of Sciences, 1978; Treshow, 1970; U.S. Environmental Protection Agency, 1973, 1978a; Van Haut and Stratmann, 1970) and, to a lesser extent, of particulate matter (Guderian, 1980) exist in the literature. Additional reports of SO_2 effects on plants have also been prepared for use in diagnostic situations (Davis, 1972a,b; Lacasse and Moroz, 1969; U.S. Environmental Protection Agency, 1976).

This chapter addresses factors that influence our ability to determine relationships between pollutant concentration and plant responses. Principal focus is placed on specifying concentrations of SO_2 and particles associated with vegetation responses ranging from the biochemical level to that of plant populations. In this process, information of historical interest has been kept to a minimum and emphasis has been placed on more recent studies that have employed modern monitoring, experimental, and statistical techniques. Studies in which SO_2 or particulate matter were determined to be the major cause of measured effects have been emphasized.

As a backdrop against which to consider pollutant effects on plants, it is important to recognize that the many factors that play a major role in determining whether a given quantity of pollutant will produce a predictable level of effect vary tremendously in nature. These factors include the type of exposure (acute or chronic), influences of stress from other biotic (insects, disease) or abiotic (edaphic or climatic) factors, the type of response measured, and the species or population under study. These factors and associated terminology have been addressed in Sections 8.2 and 8.4 for SO_2 and Sections 8.6 and 8.7 for particulate matter.

While a broad variety of responses measured following exposure of vegetation to SO_2 or particulate matter are discussed, it should be noted that not all responses are detrimental, and that not all short-term detrimental responses ultimately result in effects detrimental to plant growth and development. These concepts are developed more fully in Sections 8.2.7 and 8.3 below.

The end-point of this presentation of concepts, components, and modifiers of pollutant dose and plant response can be found in attempts to define exposure-response relationships for SO_2 and PM effects on vegetation. This is done in Sections 8.3 and 8.8 for SO_2 and particulate matter, respectively.

The concluding section on ecosystem responses (Section 8.10) derives information from a much more limited number of studies. Here reliance on the broadly based concepts of ecosystem analyses forms the basis for strong inference rather than proof of effects on a more subtle scale. In this area definitive data to evaluate the degree or extent of ecosystem changes over broad regions do not presently exist.

The following definitions have generally been acceptable to plant scientists working on air pollutant-induced effects on plants (American Phytopathological Society, 1974) and are relevant to information discussed below in this chapter:

1. Injury - a change in the appearance and/or function of a plant that is deleterious to the plant.
2. Acute Injury (effects) - injury, usually involving necrosis, which develops within several hours to a few days after short-time exposure to a pollutant, and is expressed as fleck, scorch, bifacial necrosis, etc.
3. Chronic Injury (effects) - injury which develops only after long-term or repeated exposure to an air pollutant and is expressed as chlorosis, bronzing, premature senescence, reduced growth, etc.; can include necrosis.
4. Damage - a measure of decrease in economic or aesthetic value resulting from plant injury by pollutants.

8.2 REACTION OF PLANTS TO SULFUR DIOXIDE EXPOSURES

8.2.1 Introduction

The response of plants to SO_2 exposure is a complex process that involves not only the pollutant concentration and duration of exposure but also the genetic composition of the plant and the environmental factors under which exposure occurs. In simplistic overview, this process involves entrance of SO_2 into the plant through leaf openings called stomata, and contact within the leaf with wet cellular membranes and subsequent liquid phase reactions resulting in the formation of sulfite and sulfate compounds. The formation of these compounds can initiate changes within the plant metabolic systems that will produce physiological dysfunctions. If sufficient physiological modifications occur, plant homeostasis or equilibrium is disturbed and visible symptoms of injury may or may not be manifested. Plant repair mechanisms can result in a return to homeostasis and recovery; however, should exposure occur again prior to complete recovery, further injury may occur and plant recovery is less probable.

Several plant responses to exposure to SO_2 and related sulfur compounds are possible: (1) fertilizer effects appearing as increased growth and yield; (2) no detectable responses; (3) injury manifested as growth and yield reductions without visible symptom expressions on the foliage or with only very mild foliar symptoms that are difficult to attribute to air pollution without comparing them to a control set of plants grown in pollution-free conditions; (4) injury exhibited as chronic or acute symptoms on foliage with or without associated reductions in growth and yield; and (5) death of plants and plant communities.

In some instances, the addition of low SO_2 concentrations to a plant's environment may induce a fertilizer-like response, but relatively few studies of this phenomenon on agronomic crops have been completed to date; none have shown beneficial effects on natural ecosystems. In view of the scanty data on the subject, only limited consideration of potential beneficial effects of SO_2 exposure can be undertaken at this time.

Plant injury or death may result from continued exposure to high or low pollutant doses. If such is the case, other mitigating factors may also be involved (e.g., abiotic agents or biotic disease-inducing agents such as insects). Depending upon the plant species, exact conditions of the seasonal stage of crop growth, pollutant dose, and environmental conditions, many forms of injury may take place and their relative impact may vary. Symptoms of acute and chronic injury may occur on a given plant simultaneously. Here, injury does not necessarily imply damage (i.e., economic loss). Also, the timing of pollutant exposure in relation to the physiological stage of crop development often determines the relationship of foliar injury to subsequent yield losses.

Before discussing the different types of effects of SO_2 on vegetation, certain background information is provided concerning deposition of SO_2 on plant surfaces, entrance into the plant, and distribution and reaction within the plant after SO_2 entry into plant tissues.

8.2.2 Wet and Dry Deposition of Sulfur Compounds on Leaf Surfaces

Deposition processes limit the lifetime of sulfur compounds in the atmosphere, control the distance traveled before deposition, and limit the atmospheric pollutant concentrations (Garland, 1978). A fuller discussion of pollutant transport, transformation and dry deposition is given in Chapter 6.

There have been several studies of the deposition of particulate material to natural surfaces (Chamberlain, 1975; Little and Wiffen, 1977; Sehmel and Hodgson, 1974). Very large particles are chiefly deposited by sedimentation. Particles in the range of 1 to 100 µm are also borne towards the surface by turbulence where sedimentation is supplemental to impaction on rough surfaces. Submicrometer-sized particles (e.g., sulfuric acid aerosols) and gases, including SO_2, diffuse by Brownian motion through the thin laminar layers of air close to the plant surface. This may be followed by active uptake by plants. The mean SO_2 deposition velocities are surprisingly similar for a wide range of plant leaf surfaces (Garland, 1978). (Wet deposition is discussed in Chapter 7.)

Dry deposition results in the removal of significant amounts of the larger particles from the atmosphere within 2 or 3 days following emission, but several weeks are required to remove the submicrometer fraction.

8.2.3 Routes and Methods of Entry Into the Plant

Stomata of leaves have been demonstrated to be the major avenues of SO_2 entrance into plants. Although this is a widely accepted conclusion that has been presented in numerous reviews (Guderian, 1977; Katz, 1949; Thomas and Hendricks, 1956; U.S. EPA, 1973), there is still controversy as to the importance of stomatal movement relative to plant biochemistry in determining plant sensitivity. Many factors that govern the mechanism of stomatal opening and closing have been determined to be independent of SO_2 concentrations to which a plant is exposed. Physical factors such as light, leaf surface moisture, relative humidity, and soil moisture availability influence stomatal opening and closing and play a major role in plant sensitivity by limiting passive entry of SO_2 into the leaf (Domes, 1971; Meidner and Mansfield, 1968; Setterstrom and Zimmerman, 1939; Spedding, 1969; McLaughlin and Taylor, 1981). These factors must therefore be considered when determining plant sensitivity or tolerance to entry of SO_2.

Internal resistances to flux of gases into plant leaves may also be substantial and may exceed those imposed by stomata under some conditions. Barton et al. (1980) found that photosynthetic depression in kidney beans (Phaseolus vulgaris) during SO_2 exposure was explained primarily by increases in mesophyll resistance. Stomatal resistances changed only slightly and were a minor component of total leaf resistance to CO_2 at both high (71 percent) and low (33 percent) relative humidity. Winner and Mooney (1980) also found that differences in both stomatal and nonstomatal components of leaf resistance to SO_2 uptake were associated with differences in resistance of deciduous and evergreen shrubs to SO_2.

The rate of absorption of SO_2 by plants varies not only among species, but also is influenced by previous exposures to SO_2. Rates of SO_2 absorption and of translocation of absorbed sulfur were determined in sugar maple (Acer saccharum Marsh.), big toothed aspen (Populus grandidentata Michx), white ash (Fraxinus americana L.) and yellow birch seedlings (Betula alleghaniensis Britton Betula lutea Michx. f.). Bigtooth aspen, a species sensitive to SO_2, had the highest absorption rate with no prefumigation and sugar maple, a tolerant species, had the lowest. The rate of sulfur absorption was reduced in all species except white ash, a species of intermediate sensitivity, after prefumigation with SO_2 at 1970 $\mu g/m^3$ (0.75 ppm) for 20 to 36 hrs. (Jensen and Kozlowski, 1975). The relationship between species sensitivity and absorption rate thus changed with the prefumigation treatment. The sulfur content of foliage in all species increased with SO_2 fumigation. Eight days after fumigation with $^{35}SO_2$, varying amounts of the labelled sulfur were translocated throughout the plants, including roots (Jensen and Kozlowski, 1975). Subsequent effects were not indicated.

Sulfur dioxide has been shown both to increase and decrease stomatal resistance and thus affect potential photosynthetic performance (Hallgren, 1978). Sulfur dioxide induced the closure of stomata in the Pelargonium hybrid, "Pelargonium x hortorum", especially when they had been fully opened, and necrosis was not averted (Bonte et al., 1975). Kodata and Inoue (1972) demonstrated that SO_2 entered leaves of Pinus resinosa (red pine) through stomata and accumulated in the cells around stomata for some time before diffusing inward through the leaf; i.e., internal diffusion was slower than diffusion into the leaves.

Once SO_2 has entered a leaf, it may induce stomata to remain open for longer periods of time or to open wider than before fumigation. However, exposure of plants to SO_2 (1360 $\mu g/m^3$, 0.5 ppm) at relative humidities above 40 percent caused an increase in stomatal opening (Majernik and Mansfield, 1970; Mansfield and Majernik, 1970). A 3-minute fumigation with 6550 $\mu g/m^3$ (2.5 ppm) SO_2 increased carbon dioxide uptake and stomatal opening in white mustard Brassica hirta Moench (Sinapsis alba B + B) plants. However, with the same concentration, suppressed carbon dioxide uptake and stomatal closure have also been noted (Buron and Cornic, 1973).

8.2.4 Cellular and Biochemical Changes

Based on the available literature, it is difficult to assess the relationship of SO_2-induced biochemical and/or physiological changes at the cellular level to subsequent effects on photosynthetic activity or resultant growth and yield. Numerous studies have utilized detached leaves and/or isolated chloroplasts in culture solutions for evaluation of biochemical or physiological effects, but their use for field estimations under ambient conditions remains limited.

Recent studies have also shown a variety of SO_2-induced biochemical effects: enzyme inhibition (Pahlich, 1971, 1973; Ziegler, 1972); interference with respiration (Haisman, 1974); interference with energy transduction (Ballentyne, 1973); interference with lipid biosynthesis (Malhotra and Kahn, 1978); alterations in amino acid content and quality (Godzik and Linskens, 1974); and chlorophyll loss (Rao and LeBlanc, 1965). Pahlich (1975) has rationalized some of this diverse list of effects in terms of sulfite and sulfate accumulation by exposed plant tissues.

Vogl et al. (1965) attempted to integrate biochemical responses with the type and magnitude of resultant plant effects (Table 8-1). In Table 8-1 degrees of injury are classified from A to E, i.e., from no visible injury to "catastrophic" in Vogls' terminology. Development of models is necessary to relate changes in physiology (biochemical responses) of particular plant species to altered growth and productivity, both qualitatively and quantitatively. That is, the relationship between biochemical changes or responses and visible injury at various SO_2 exposure levels remains to be better defined.

Horsman and Wellburn (1976) have prepared an extensive listing of reported metabolic or enzymatic effects of SO_2 on plants or plant tissues. In only one of eleven studies reviewed was an increase in photosynthesis ($^{14}CO_2$ fixation) noted as a positive or beneficial effect in response to exposure to SO_2 or its derivatives; the remaining effects observed were negative or detrimental.

With SO_2, which upon absorption is hydrolized to $SO_3^=$ and then to $SO_4^=$ and subsequently incorporated into S-containing amino acids and proteins, the rate of entry is particularly important for determining toxicity. Plants have an inherent, and apparently species-dependent, capacity to absorb, detoxify, and metabolically incorporate SO_2 and may absorb low concentrations of SO_2 over long time periods without damage. Thomas et al. (1943), for example, exposed alfalfa to SO_2 continuously at 520 µg/m^3 (0.20 ppm) for 8 weeks without adverse effects. Toxicity to SO_2 may occur during short episodes when the SO_2 to $SO_4^=$ conversion rate is exceeded and the extremely toxic sulfite ($SO_3^=$) form accumulates (Ziegler, 1975). During longer exposures at lower SO_2 concentrations, $SO_4^=$ may accumulate as the capacity for metabolic incorporation of $SO_4^=$ is exceeded, and chronic symptoms may appear. It is therefore reasonable to expect that either no effects or beneficial effects may be associated with low-level SO_2 exposures; but detrimental (even fatal) toxic effects may occur at increasingly higher SO_2 ex-

TABLE 8.1. RELATIONSHIP OF BIOCHEMICAL RESPONSE TO VISUAL SYMPTOMS OF PLANT INJURY

Degree of injury	Description of Injury					
			Injury to:		Ability to recover in:	
	Visible symptoms	Symptoms of biochemical injury in leaf cells	Assimilation organs	Whole plant	Assimilation organs	Whole plant
A	none	stress on buffer systems	none	none		
B	not detectable	photosynthesis adversely affected, diminished assimilation rate	temporary impedance of gaseous exchange	not detectable	very quick, completely	
C	loss of assimilation capacity through:	diminished activity of enzymes	prolonged impedance of gaseous exchange	reduced growth (deficiency conditions)	slowly, completely	slowly, completely for perennials
	1) premature death of assimilation organs (leaves, needles)	effect upon chlorophyll				
	2) diminished growth of new tissues (shorter needles, etc.)					
D	necrosis of the assimilating and active plant tissues	death of cells through protein and enzyme degradation	irreversible injury: necrosis of some assimilation organs or parts thereof	loss of assimilation capability	quick, not completely, sometimes (for isolated tissues) not any more	slowly, completely for perennials
E	destruction of all important assimilatory plant tissues	death of organs	irreversible injury to all assimilation organs	destruction of assimilation capability	not any more	sometimes (for isolated tissues)

Vogl, et al., 1965.

posure levels once the capacity for conversion of $SO_3^=$ to $SO_4^=$ and the transformation of $SO_4^=$ in the leaf are exceeded.

8.2.5 Beneficial "Fertilizer" Effects

Under certain conditions, atmospheric SO_2 can have beneficial effects on agronomic vegetation (Noggle and Jones, 1979). Sulfur is one of the elements required for plant growth and Coleman (1966) reported that crop deficiencies of sulfur have been occurring with increasing frequency throughout the world. The sulfur required to maintain high crop production ranges from 10 to 40 kg/ha per year. Figure 8-1 presents a map of sulfur-deficient soils of the United States (The Sulphur Institute, 1979).

Cowling et al. (1973) found beneficial effects of SO_2, such as increases in yield and sulfur content, in perennial ryegrass that was grown with an inadequate supply of sulfur to the roots. Faller (1970) conducted a series of experiments to determine effects of varying atmospheric concentrations of SO_2 on sunflower, corn, and tobacco. In these studies, plants were grown in nutrient media containing adequate supplies of all essential elements except sulfur, which was low. Plants grown in the atmosphere without SO_2 developed sulfur-deficiency symptoms within a few days. In other treatments, total plant yield increased to some extent when increasing concentrations of SO_2 were added to the atmosphere during plant growth. For tobacco, the total dry weight increased by up to 40 percent. Yields of leaves and stems alone increased by 80 percent while dry weights of tobacco increased even at the highest SO_2 concentration used (1490 μg/m^3, 0.57 ppm); sunflower and corn had their highest biomass at SO_2 concentrations of 1050 μg/m^3 (0.40 ppm) and 520 μg/m^3 (0.20 ppm), respectively. Beyond these concentrations, visible injury was observed. Additional studies by Faller (1970) with ^{35}S suggest that up to 90 percent of plant sulfur requirements may originate from the atmosphere under the specific experimental conditions.

No monitoring or handling procedures for SO_2 delivery were presented in a study by Thomas et al. (1943); however, its results indicated that SO_2 could serve as a source of nutrient S when alfalfa, grown in sulfur-deficient solutions, was fumigated with approximately 260 μg/m^3 (0.10 ppm) SO_2 for 6 to 7 hr/day, 6 days/week for the growing season, but had little effect when the growth medium contained sufficient sulfur.

Noggle and Jones (1979) reported the results of a 2-year study comparing the contributions of soil and atmospheric sulfur to the sulfur requirements of cotton and fescue when the plants were potted in soil and ^{35}S was added as the S nutrient. Cotton was more efficient than fescue in accumulating sulfur from the atmosphere. The amount of sulfur accumulated from the atmosphere was apparently influenced by the amount of sulfur supply in the soil relative to the sulfur requirements of the plant. A crop grown in a sulfur-deficient soil will accumulate more sulfur from the atmosphere than the same crop grown in a soil that has an adequate supply of sulfur. Noggle and Jones (1979) also showed that cotton grown in specifically designed growth containers in the vicinity of certain coal-fired power plants accumulated significant amounts of atmospheric sulfur (as SO_2) and produced significantly more biomass than

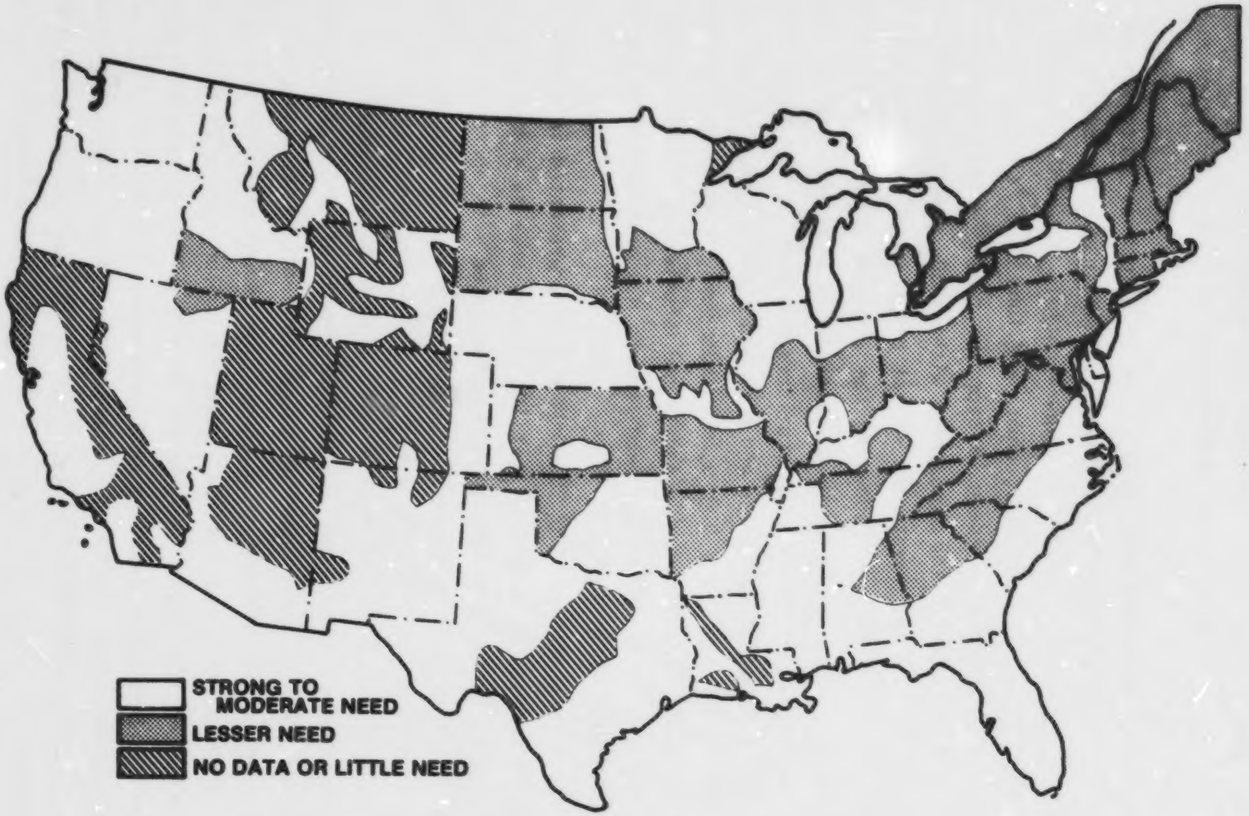

Figure 8-1. Map of the United States indicating major areas of sulfur-deficient soils.

Source: The Sulphur Institute (1979).

those grown at a location farther from the industrial source of sulfur. Thus, under appropriate conditions, such as with sulfur-deficient soils, the atmosphere can be an important source of sulfur for plant requirements. It should be noted, however, that SO_2 monitoring data were not presented in the report.

Similar stimulatory responses of Oryzopsis hymenoides (a desert grass) were noted following continuous exposure to SO_2 concentrations of 80 $\mu g/m^3$ (0.03 ppm) and 160 $\mu g/m^3$ (0.06 ppm) for 6 weeks (Ferenbaugh, 1978). The noted increases in productivity expressed as mg dry wt/plant were not statistically different from the control plants, but the trends were stable for these doses. Exposure to 340 $\mu g/m^3$ (0.13 ppm) and 660 $\mu g/m^3$ (0.25 ppm) SO_2 under identical conditions however, resulted in foliar symptoms and decreased productivity.

Studies that have demonstrated beneficial effects, as well as those demonstrating detrimental effects, must be evaluated in terms of their single influence on one crop. There is little doubt that direct application of sulfur as a nutrient to certain crops grown under borderline or sulfur-deficient conditions may result in increased productivity for that crop. Plants normally take up sulfur in the form of sulfate via the roots; however, they can also utilize SO_2 taken up through the leaf stomata. Faller (1971) has shown that tobacco plants can utilize SO_2 as a sulfur source in deficient soils. Studies of this type (Faller, 1970) have led some individuals to conclude that SO_2 emitted from industrial sources is actually beneficial as an aerial fertilizer for plants growing in sulfur deficient soils (Terman, 1978; Noggle and Jones, 1979). The situation is not as simple as portrayed. A small amount of atmospheric SO_2 can be nutritionally beneficial to plants in the short term. However, in the long term, large, uncontrolled, and frequent applications of sulfur compounds such as SO_2 can be detrimental. Not all plant species have the same nutritional requirements. In addition, the rate at which plant species assimilate sulfur is influenced by such variables as the physiological status of the plants, age and time during the growing season when the application occurs, availability of soil nutrients, and light intensity. When more sulfur is available than can be assimilated by plants, it is accumulated in the leaf tissue (Ulrich et al., 1967; Legge et al., 1976; Cowling and Koziol, 1978; Thompson and Katz, 1978). Sulfur accumulation in the leaf can reach toxic levels and adversely affect plant growth (Katz, 1949; Linzon et al., 1979). Growth rates of leaf tissues, available nutrient supplies, and environmental factors affecting stomatal opening and closing should all be considered as influencing the rate of sulfur accumulation in plant tissues (Bell and Clough, 1973).

The effects of nutrient forms of nitrogen being delivered to the plant along with SO_2 in plumes under field conditions have not been investigated. However, Cowling and Lockyer (1978) demonstrated that ryegrass (the S 23 cultivar), when grown under sulfur (S) deficiency and at low nitrogen (N), did not respond to 50 $\mu g/m^3$ (0.02 ppm) SO_2 for 85 days, whereas plants grown at high N under the same exposure conditions responded with a 227 percent yield increase and S-deficiency symptoms were alleviated. Data presented previously by Cowling and Jones (1971) showed that at high levels of N and inadequate levels of S, nitrate-nitrogen accumulated in ryegrass, thus indicating that protein synthesis was inhibited. Long-term natural ecosystem studies showing the beneficial effects of SO_2 deposition are not available, whereas those showing detrimental effects are available (Legge et al., 1981; Legge, 1980; Winner et al., 1978; Winner and Bewley, 1978a). Since agronomic and natural ecosystems are often physically proximal to one another, further research on the potential influence of sulfur compounds to each singly and collectively is greatly warranted.

The Sulphur Institute (1971) published a review of crop response to sulfur; the reader is referred to that review for more information on beneficial effects of sulfur on plants.

8.2.6 Acute Foliar Injury

This type of injury occurs following rapid absorption of a toxic dose of SO_2 and results at first in marginal and intervenal areas having a dark green, watersoaked appearance. After desiccation and bleaching of tissues, affected areas become light ivory to white in most broadleaf plants. Some species show darker colors (brown or red), but an exact line of demarcation characteristically exists between symptomatic and asymptomatic portions of leaf tissues. Bifacial necrosis is common. In monocotyledons (corn, grasses), foliar injury occurs at the tips and in lengthwise strips along parallel veins (U.S. EPA, 1976).

In conifers, acute injury on foliage usually appears as a bright orange red tip necrosis on current-year needles, often with a sharp line of demarcation between the injured tips and the normally green bases. Occasionally, injury may occur as bands at the tip, middle, or base of the needles (Linzon, 1972). Recently incurred injury is light colored; but later, bright orange or red colors are typical for the banded areas and tips. As needle tips die, they become brittle and break, or whole needles drop from the tree. Pine needles are most sensitive to SO_2 during the period of rapid needle elongation, but injury may also occur on mature needles (Davis, 1972a).

8.2.7 Chronic Foliar Injury

Visible plant injury not involving collapse and necrosis of tissues is termed chronic injury. This type of visible injury is usually the result of variable fumigations consisting of either short-term, high-concentration or long-term, low-concentration exposures to SO_2. It has also been called "sulfate injury" since a slow accumulation of sulfate results from such exposures (Daines, 1968). Within substomatal cavities, SO_2 reacts quickly with intercellular water to form sulfite and bisulfite. These substances are slowly oxidized to sulfate which is approximately one-thirtieth as toxic as sulfite and bisulfite (Thomas, 1951). If the capacity of plant tissues to convert sulfite and bisulfite to sulfate is not exceeded or sulfate elimination processes are not overwhelmed, visible expression of symptoms will not occur. However, as sulfite and bisulfite ions are formed and as sulfate accumulates to phytotoxic levels, then chronic symptoms first appear as various forms of chlorotic (yellowing) patterns. As sulfite and bisulfite ions continue to accumulate, destruction of individual chloroplast membranes or reduction of chlorophyll production ensues, resulting in reddening or bleaching of cells without necrosis (Thomas, 1951). Following such accumulations, a fine gradation occurs between chronic and acute symptom expressions. Sulfate levels in plants ambiently exposed to SO_2 have been shown to be several times greater than those in control plants (Linzon, 1958).

In broadleaf plants chronic injury is often expressed in tissues located between veins, with various forms of chlorosis predominating. Chlorotic spots or mottle may persist following exposure or may subside and disappear after pollutant removal or other changes in environmental conditions (Jacobson and Hill, 1970). Chronic effects of SO_2 in conifers are generally first expressed on older needles (Linzon, 1966). Chlorosis of tissues starting at the tips progresses down the needle towards the base, i.e., symptoms progress from the oldest to youngest tissues. Advanced symptoms may follow, involving reddening of affected tissues (Linzon, 1978).

8.2.8 Foliar Versus Whole Plant Responses

The presence of acute or chronic foliar injury is not necessarily associated with growth or yield effects. Furthermore, when present, the degree of foliar injury may not always be a reliable indicator of subsequent growth or yield effects. Yield effects in the absence of foliar injury, for example, have been reported for soybeans in field fumigation experiments (Sprugel et al., 1980; Heagle et al. 1974) and in chamber exposures (Reinert and Weber, 1980). Sprugel et al. (1980), utilizing the Zonal Air Pollution System (ZAPS), reported significant yield reductions in Wells soybeans exposed to mean SO_2 levels of 240, 260, 500, 660 or 940 $\mu g/m^3$ (0.09, 0.10, 0.19, 0.25, or 0.36 ppm) for an average of 4.2 hr/day intermittantly for 18 days during July and August 1978, but visible leaf injury was seen only at the highest SO_2 levels. The "ZAPS" permits significant variation in pollutant concentrations. For example, in 1977, in experiments reporting a mean SO_2 concentration of 790 $\mu g/m^3$ (0.30 ppm), the actual concentration ranged from 0.00 to 3140 $\mu g/m^3$ (0.00 to 1.20 ppm) SO_2 (Miller et al., 1980). The "ZAPS" type of exposure system, however, tends to simulate ambient conditions near a point source. Under the "ZAPS" system, SO_2 or other pollutants are delivered through a system of one-inch pipes suspended about 75 cm above the soil surface in field plots of 0.5 hectares. Pollutant samplers are located in the plots to measure the amount of pollutant delivered to the vegetation. Though flow rates through the system are constant, the canopy-level concentrations vary depending upon the meteorological conditions, principally wind speed and temperature, so that concentrations are often higher near the pipes.

In a greenhouse chamber study of O_3 + SO_2 interaction, Reinert and Weber (1980), exposed Dare soybeans for 4 hr/day, 3 times per week, for 11 weeks. They reported significant growth reductions in the absence of visible injury at 660 $\mu g/m^3$ (0.25 ppm) SO_2 when the treatment sums of squares were partitioned. On the other hand, Heagle et al. (1974) also exposed Dare soybeans but found that plants exposed to 260 $\mu g/m^3$ (0.10 ppm) SO_2 6 hr/day for 133 days in closed field chambers exhibited no significant yield reductions, even in the presence of foliar injury.

Additional examples of ambiguities are found in some of the literature dealing with grasses. In a preliminary study, S 23 ryegrass exhibited significantly reduced growth when exposed to 310 μg/m^3 (0.12 ppm) SO_2 for 9 weeks or 174 μg/m^3 (0.067 ppm) SO_2 for 26 weeks (Bell and Clough, 1973); however, the only foliar injury noted was a slight chlorosis and an enhanced rate of leaf senescence. Ashenden (1978) noted similar significant growth reductions for cocksfoot grass when exposed to 290 μg/m^3 (0.11 ppm) SO_2 for [illegible] weeks; reductions ranged from 32 to 52 percent for various yield parameters while foliar necrosis was only 5 percent. On the other hand, exposure of S 23 ryegrass to 50 or 370 μg/m^3 (0.02 or 0.14 ppm) SO_2 in two successive growth periods of 29 and 22 days was reported to result in foliar injury at the high concentration, but no yield effects at either concentration (Cowling and Koziol, 1978). Neither net photosynthesis nor dark respiration was significantly affected.

Different plant species differ in tolerance to SO_2 injury. Leaf injury and radial growth were evaluated on Douglas fir and ponderosa pine growing in nursery plots exposed to various doses of SO_2 in controlled fumigations to determine their tolerance to SO_2 exposure (Katz and McCallum, 1952). Slightly injured ponderosa pine (10 percent foliar symptoms) exhibited no significant deviations in growth while slightly injured Douglas fir (10 percent foliar symptoms) showed definite growth retardation in comparison to control plants. The growth retardations were evident for 3 years after SO_2 exposure, followed by substantial recovery.

Sulfur dioxide concentrations have also been negatively correlated with annual ring width in Norway spruce (Keller, 1980). Exposures to 130, 260 and 520 μg/m^3 (0.05, 0.10, and 0.20 ppm) SO_2 were continuous for 10 weeks in the spring. Some injury was noted at 260 μg/m^3 (0.10 ppm) and 520 μg/m^3 (0.20 ppm), but a distinct decline of wood production was also found in cases where no visible injury occurred. When dormant seedlings of beech were exposed to the same SO_2 concentrations of 130, 260 and 520 μg/m^3 (0.05, 0.10, and 0.20 ppm) for about 16 weeks, there was an increase in the number of terminal buds that failed to "break" in the spring for the 260 (0.10 ppm) and 520 μg/m^3 (0.20 ppm) treatments (Keller, 1978).

The literature is ambiguous concerning SO_2-induced growth and yield effects and correlations with visible foliar injury. No studies considered all of the potential variables that could affect plant response. This is a virtual impossibility for a single study and is especially true for field studies (which are most relevant for present purposes) where many environmental variables cannot be controlled. Also, many of the studies that have demonstrated adverse effects of SO_2 at low concentrations have utilized sensitive cultivars of plant species (which may or may not be representative of plant populations as a whole) and maintained exposure conditions conducive to maximum plant sensitivity. However, from the data available, we can conclude that growth and yield effects are not necessarily related to foliar injury. Depending upon the plant affected, the environmental conditions, and the pollutant exposure conditions, one may observe yield effects without injury, injury without yield effects, or positive associations between injury and yield effects.

8.2.9 Classification of Plant Sensitivity to Sulfur Dioxide

Limitations of space do not permit a listing here of all plants known to be sensitive to various doses of SO_2. Furthermore, in a listing of sensitive plants, the evidence collected should also indicate environmental, genetic, and cultural considerations that may in fact determine such sensitivities. In addition, general descriptions are difficult because plant responses to air pollutants vary at the genus, species, variety, and cultivar levels. Some listings of plants according to relative sensitivities have been prepared, however, using as a basis the expression of visible symptoms by any given plant. Injury expressed as losses in growth or yield, however, has generally not been considered in preparing such lists.

Jacobson and Hill (1970) published a list of plants sensitive to major phytotoxic air pollutants, and Linzon (1972) has classified 36 tree species as being either tolerant, intermediate, or sensitive to SO_2. These sensitivity lists do not identify the pollutant doses required to induce visible injury on indicator species. However, Jones et al. (1974) have published such details based upon observations over a 20-year period of 120 species growing in the vicinity of coal-fired power plants in the southeastern United States (Table 8-2).

Other compilations have also been presented, including that of Davis and Wilhour (1976) which provides information on an international basis, and the report of Hill et al. (1974) for vegetation native to the southwestern deserts of the United States.

Extensive efforts have been made to identify and develop certain sensitive plant species as potential bioindicators of ambient air SO_2 effects. Perhaps the most extensively examined plants for this use are eastern white pine (Pinus strobus) and numerous species of lichens. The literature on white pine has been reviewed by Gerhold (1977), and the most recent review discussing lichens as bioindicators was prepared by LeBlanc and Rao (1975). Other reports mention the use of various ornamentals (Daessler et al., 1972; Heggestad et al., 1973; Pelz, 1962); bluegrass cultivars (Murray et al., 1975); scotch pine (Demeritt et al., 1971); hybrid poplar (Dochinger and Jensen, 1975); and trembling aspen (Karnosky, 1977).

Bioindicators for determining SO_2 effects must be used with caution, however, because other factors such as drought, nutrient imbalances, and other pollutants may induce injury symptoms that mimic those of SO_2. Therefore, several bioindicators should be used if any given geographic area is to be evaluated for possible SO_2 effects. In addition, individual species and more complicated plant bioindicator systems are not as effective in detecting SO_2 at low concentrations as are sophisticated instruments.

8.3 EXPOSURE-RESPONSE RELATIONSHIPS - SULFUR DIOXIDE

The primary focus of exposure-response studies should be to delineate relationships between measurable indices of exposure and meaningful parameters of plant response. This section will examine such relationships both from the perspective of defining empirically demonstrated exposure-response (or exposure-effect) relationships based on our present knowledge and from the perspective of what generalizations might be reasonably made concerning likely effects of SO_2 on plants under ambient conditions.

TABLE 8-2. SENSITIVITY GROUPINGS OF VEGETATION BASED ON VISIBLE INJURY AT DIFFERENT SO_2 EXPOSURES[a]

Sensitivity Grouping	SO_2 concentration, μg/m^3 (ppm), and duration time, hr			Plants
	Peak[b]	1-hr	3-hr	
Sensitive:	2620-3930 μg/m^3 (1.0 - 1.5 ppm)	1310-2620 μg/m^3 (0.5 - 1.0 ppm)	790-1570 μg/m^3 (0.3 - 0.6 ppm)	Ragweeds Legumes Blackberry Southern pines Red and black oaks White ash Sumacs
Intermediate:	3930-5240 μg/m^3 (1.5 - 2.0 ppm)	2620-5240 μg/m^3 (1.0 - 2.0 ppm)	1570-2100 μg/m^3 (0.6 - 0.8 ppm)	Maples Locust Sweetgum Cherry Elms Tuliptree Many crop and garden species
Resistant:	>5240 μg/m^3 (> 2.0 ppm)	>5240 μg/m^3 (> 2.0 ppm)	>2100 μg/m^3 (> 0.8 ppm)	White oaks Potato Upland cotton Corn Dogwood Peach

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[a] Based on observations over a 20-year period of visible injury occurring on over 120 species growing in the vicinities of coal-fired power plants in the southeastern United States.

[b] Maximum 5 minute concentration.

Source: After Jones et al., 1974.

The dose of SO_2 to which vegetation may be exposed is conventionally designated as the product of the concentration of SO_2 in the plant's environment times the duration of exposure. The term exposure dose refers to the amount of pollutant at the external plant surface. Response may be characterized by a measurable change in any plant function such as biochemical pathways, gas exchange rates, photosynthetic rates, physiological reactions, degree of visibly recognizable leaf injury, or subsequent growth and yield. Plant responses may be beneficial or detrimental (see Section 8.2.5). They may also involve the expression of growth and yield effects without foliar symptoms (see Section 8.2.8) or may lead to overt symptoms that seldom become more serious than those associated with acute injury (see Section 8.2.6) or chronic injury (Section 8.2.7).

In interpreting exposure-response studies wherein a measured plant response is correlated with the length of exposure, it is important to realize that the relationship between exposure and the amount of pollutant entering the plant may be influenced significantly by environmental factors which control rates of pollutant flux into plant leaves and by plant factors that determine the metabolic fate of the pollutant within leaf tissues as noted earlier (see Figure 8-2).

The role of short-term fluctuations in SO_2 may be particularly important where impacts of point sources are of concern (McLaughlin et al., 1976). Here concentrations may fluctuate widely during exposure, and damage to vegetation may be most closely associated with short-term averages (1 hr) or even brief peak concentrations (The term "peak" is not precise, but depends on instrumentation and the opinion of the scientist. However, a peak is usually of only a few minutes duration.) Consistent with this, laboratory experiments by Zahn (1961, 1970) have demonstrated the greater relative toxicity of short-term exposures at high concentrations of SO_2 than longer-term exposures with the same total treatment. Also, more recently McLaughlin et al. (1979) studied the effects of varying the peak to mean SO_2 concentration ratio on kidney beans in short-term (3 hr) treatments with SO_2. They found that increasing the peak to mean ratio from 1.0 (steady state treatments at 0.50 ppm for 3 hours) to 2.0 (3 hr treatments with peak = 2620 $\mu g/m^3$ or 1.0 ppm) did not alter post fumigation photosynthetic depression. Further increasing the ratio to 6.0 (1 hour exposure with peak = 5240 $\mu g/m^3$ or 2.0 ppm), however, tripled the postfumigation photosynthetic depression. Total dose delivered in the three exposures was 1.5, 1.8, and 1.1 ppm hr, respectively. Clearly the quantity of SO_2 to which the plants are exposed may have a very different effective potential as the parameters of the exposure are changed.

Another important aspect of exposure is the frequency and duration of periods of low SO_2 stress. Zahn (1970) emphasized that periods of low SO_2 concentration may be critical to the recovery potential of plant systems following exposure to elevated levels of SO_2. Thus, continuous exposure systems probably overestimate the toxicity of the delivered dose in many cases because physiological recovery is not permitted. Such recovery would be expected under

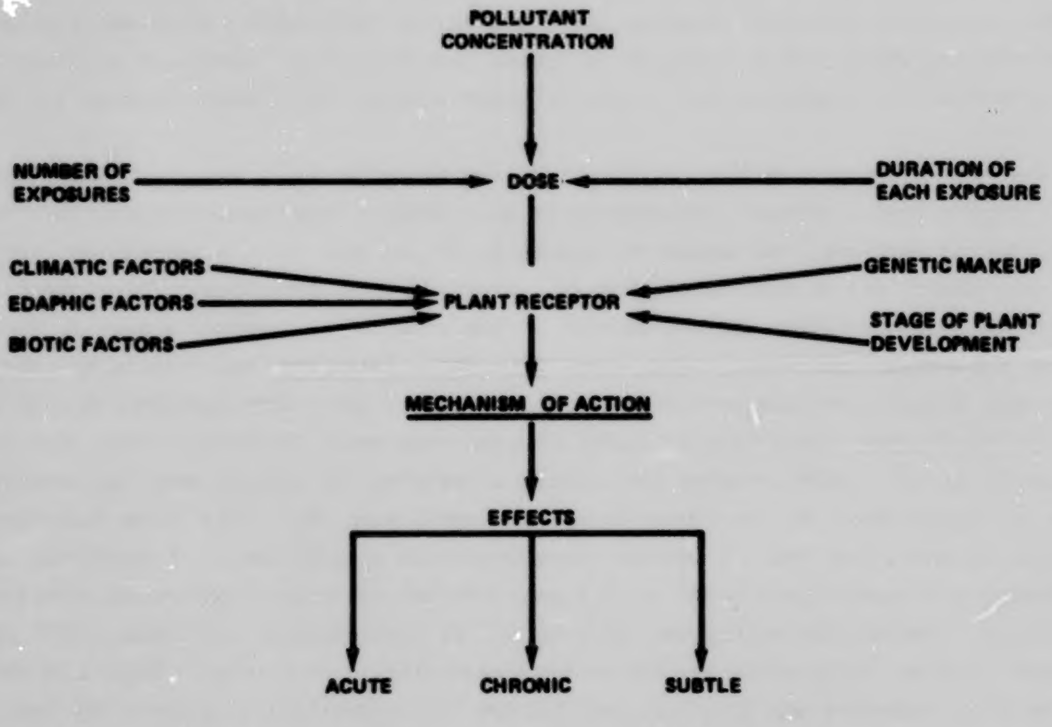

Figure 8-2. Conceptual model of the factors involved in air pollution effects (dose-response) on vegetation.

Source: Heck and Brandt, 1977.

most exposure regimes in the field where fluctuating synoptic or local meteorological conditions strongly influence exposure patterns.

Equally critical in defining the biologically significant features of exposure regimes is a description of the significant parameters of plant response. Sections 8.2.1 through 8.2.9 have emphasized many types of responses that may be elicited by exposure to SO_2. In interpreting or predicting plant response to SO_2, it is important to keep in perspective the fact that plant growth and development represent an integration of cellular and biochemical processes, just as community behavior is an integration of the performance of component species. The internal allocation of resources (carbon, water, and nutrients) to growth is an integrative and, in many cases, resilient process that plays a major role in determining how both individual plants and plant communities respond to environmental stress (McLaughlin and Shriner, 1980; see also Section 8.10). The fact that a response is measured following exposure to a given concentration of SO_2 may be of interest in understanding the mechanism of action and in identifying the biologically significant features of exposure; however, it does not necessarily mean that an effect will be measured at a subsequent higher level of plant organization. Responses at higher levels of plant organization, however, must be viewed within the perspective of the increasingly complex biotic and abiotic factors that control plant response (see Section 8.5) and must influence our attempts to move the focus of our studies from processes to plants and from plants to communities. These factors similarly limit our capacity to extend experimental protocols beyond the confines of our carefully controlled laboratory studies to more natural, and more variable, field situations.

Several attempts have been made to characterize dose-response relationships in a mathematical sense using monitored concentrations, duration times, and injury thresholds as modified by physical and biotic factors expressed as constants (O'Gara, 1922; Thomas and Hill, 1935; Zahn, 1963a,b). However, their consistency and usefulness are limited by numerous physical and biotic factors that must be considered in evaluating dose-response data. Changes in exposure conditions, differences in exposure methodology and efficiency of monitoring equipment, and consistency of measurements within a study and between studies on the same plant directly influence results.

Data regarding SO_2 effects on plant growth and yield in most cases provide the most relevant basis for studying dose-response relationships. As a whole-plant measurement, plant productivity is an integrative parameter that considers the net effect of multiple factors over time. Productivity data are presently available for a wide range of species under a broad range of experimental conditions. However, results are neither necessarily closely comparable across sometimes very divergent experimental conditions, nor are all necessarily useful for criteria development purposes. Consequently, rather than review all such studies here in detail, summaries of their data have been tabulated separately in Appendix 8A for: controlled field exposures (Table 8A-1); laboratory studies with agronomic and horticultural crops (Table 8A-2) and tree species (Table 8A-3); and a variety of studies with native plants (Table 8A-4).

Only the most important or salient points for present purposes will be discussed here, and studies of SO_2 exposure effects on plants under field conditions will be emphasized.

Relatively few crops of economic importance have been studied under field conditions. Results obtained for SO_2 exposure effects on yields for certain commercially important species are summarized in Table 8-3 (statistical significances are noted in the table). Of the results listed, those for soybeans, wheat and barley are economically the most important. For soybeans, single 4.33 hr exposures to 3670, 4450, or 5240 $\mu g/m^3$ (1.4, 1.7, or 2.0 ppm) SO_2 induced 4.5, 11, and 15 percent yield decreases, respectively. Also, 5.2 to 15 percent yield losses in soybean cv Wells were observed with intermittent exposures to 240 to 940 $\mu g/m^3$ (0.09 to 0.36 ppm) SO_2 for an average fumigation period of 4.2 hours on 18 different days scattered from July 19 through August 27 of the soybean growing season (Sprugel et al., 1980). A greater yield loss (20.5 to 45 percent) resulted when soybeans were exposed for 4.7 hours to SO_2 at 790, or 2070 $\mu g/m^3$ (0.30 or 0.79 ppm) on 24 different days scattered through the soybean growing season using the ZAPS exposure approach in an attempt to approximate ambient conditions. The latter of the intermittently repeated exposures, therefore, appeared to have a detrimental yield effect as great or greater than single exposures of 5240 $\mu g/m^3$ (2.0 ppm) for 4 1/3 hours.

In contrast to effects reported by Sprugel et al. (1980) at 240 and 260 $\mu g/m^3$ (0.09 and 0.10 ppm) SO_2, Heagle et al. (1974) exposed the soybean cv. Dare in closed-top chambers in the field for 6 hours per day at 260 $\mu g/m^3$ (0.1 ppm) SO_2 for 133 days and noted little effect other than defoliation. The contrasting results may be due to cultivar (Wells vs. Dare) differences. Or, as Sprugel et al. (1980) suggest, the difference in results could be due to differences in the exposure systems. The closed field chamber of Heagle et al. may have modified the macroenvironment of the plants being exposed, whereas with the open-air ZAPS system, occasional high peak exposures may result. One such peak of 2100 $\mu g/m^3$ (0.8 ppm) SO_2 for one 2-minute sample was measured in 1977. Thus, the reductions in yield seen in the Sprugel et al. (1980) studies could have been due to the peaks rather than the general mean. Peaks of this type, however, are not unusual but rather are typical of air pollution episodes, particularly near point sources. Another possible factor is that in the ambient air there is a possibility of synergistic interactions. Such interactions with ozone may have increased the effects of SO_2 in the Sprugel et al. (1980) study. In 1977, ozone concentrations in the field were monitored during the first part of the season and averaged between 130 and 160 $\mu g/m^3$ (0.05 and 0.06 ppm) during the fumigations and, on at least two occasions, the concentrations exceeded 260 $\mu g/m^3$ (0.1 ppm) for one hour or more. Sprugel et al. (1980), however, point out that the study of Heagle et al. (1974) produced no evidence for greater-than-additive effects of SO_2/O_3 on soybeans. Again, the conditions in the field under which the soybeans were exposed using the ZAPS system probably more nearly simulate those under which field-grown crops would normally be exposed, and the ZAPS results rather clearly indicate that repeated 4.7 hr exposures to $\geq$790 $\mu g/m^3$ ($\geq$0.30 ppm) SO_2 during the growing season markedly reduced (by >20 percent) soybean yields under the conditions prevailing during the study.

As for other commercially important crop species, neither spring wheat or Durum wheat exhibited significant yield loss effects when exposed to intermittent concentrations of SO_2 from 660 (0.25 ppm) up to 3140 $\mu g/m^3$ (1.2 ppm), 3 times or 7 times over the growing season. When concentrations of 80 (0.03 ppm) to 390 $\mu g/m^3$ (0.15 ppm) for 72 hours per week were used for the growing season these intermittent exposures resulted in no significant yield loss effects. Barley responded in the same manner as Durum wheat, showing no significant response at any of the SO_2 concentrations or exposure times. Similar statements can be made regarding the results of repeated 3 hr 660 (0.25 ppm) to 3140 $\mu g/m^3$ (1.20 ppm) SO_2 exposures of alfalfa noted in Table 8-3 (Wilhour et al., 1978).

Table 8A-2 (Appendix 8A) presents a summary of studies that investigated effects of SO_2 on agronomic and horticultural crops grown and fumigated within artificial exposure-chamber or growth-chamber systems rather than grown in typical field soils. It is difficult to determine the significance of the results of such studies in relation to similar fumigations under field conditions. Additionally, with the exception of a relatively few studies, the doses used for exposure treatments would be considered in excess of those expected under ambient field conditions.

A few major investigations of the effects of SO_2 on vegetational species growing under natural conditions have been reported (Linzon, 1971, 1980; Dreisinger, 1965; Dreisinger and McGovern, 1970; Materna et al., 1969; Vins and Mrkva, 1973; Legge, 1980; Legge et al., 1981). Table 8-4 illustrates the degree of injury to eastern white pines (Pinus strobus) observed over a 10-year period (1953-1963) in the sulfur-fume-effects area of smelters near Sudbury, Ontario, Canada (Linzon, 1980). Linzon (1971, 1978) has indicated that a pollution (SO_2) gradient existed within the designated study area, and effects correlated well with this gradient. Chronic effects on forest growth were prominent where SO_2 air concentrations annually averaged 45 or 115 $\mu g/m^3$ (0.017 or 0.045 ppm) SO_2, and such effects were not reported in areas receiving 21 $\mu g/m^3$ (0.008 ppm) SO_2 annually. Although monitoring of SO_2 was conducted in these studies, neither concentrations of other air pollutants nor their potential effects were evaluated by the authors. It is also not clear to what extent high, short-term, peak SO_2 exposures may have contributed to the effects reported to be associated with these annual average SO_2 concentrations.

Ambient air exposures to sulfur dioxide that caused injury to a variety of both sensitive and non-sensitive species of vegetation are shown in Table 8-5. The data listed in the table were derived from the work of Dreisinger and McGovern (1970) and has been previously reviewed and cited in EPA's Revised Chapter 5 for Air Quality Criteria for Sulfur Oxides (U.S. EPA, 1973) and the National Academy of Sciences report on sulfur oxides (NAS, 1978). In addition, air quality data from the work have been analyzed and discussed by Linzon (1971). As presented and concluded by Dreisinger and McGovern in their work and cited in the NAS report,

TABLE 8-3. EFFECTS OF EXPOSURE TO SO_2 ON PLANTS UNDER FIELD CONDITIONS[a]

Plant species	Type of exposure	Time	Concentration µg/m³	(ppm)	Effects	References
Soybeans	P	4 1/3 hrs	3670	(1.4)	4.5% loss in yield (N.S.)	Miller et al. 1979
Soybeans	P	4 1/3 hrs	4450	(1.7)	11% loss in yield**	
Soybeans	P	4 1/3 hrs	5240	(2.0)	15% loss in yield*	
Soybeans cv Wells	I	4.2 hrs × 18 exposures	240	(0.09)	6.4% yield loss**	Sprugel et al. 1980 and Miller et al. 1980
Soybeans cv Wells	I	4.2 hrs × 18 exposures	260	(0.10)	5.2% yield loss*	
Soybeans cv Wells	I	4.2 hrs × 18 exposures	500	(0.19)	12.2% yield loss**	
Soybeans cv Wells	I	4.2 hrs × 18 exposures	660	(0.25)	19.2% yield loss**	
Soybeans cv Wells	I	4.2 hrs × 18 exposures	940	(0.36)	15.9% yield loss**	
Soybeans cv Wells	I	4.7 hrs × 24 exposures	310	(0.12)	12.3% yield loss**	
Soybeans cv Wells	I	4.7 hrs × 24 exposures	790	(0.30)	20.5% yield loss**	
Soybeans cv Wells	I	4.7 hrs × 24 exposures	2070	(0.79)	45.3% yield loss**	
Soybeans cv Dare	I	6 hrs/day for 43-92 and 133 days	260	(0.10)	No effect until 92nd day, 12% defoliation (N.S.)	Heagle et al. 1974
Wheat	I	3 hrs × 8 exposures/growing season	100	(0.04)	No yield effect	Sij et al. 1974
Wheat	I	3 hrs × 7 exposures/growing season	1180	(0.45)	No yield effect	
Spring Wheat	I	3 hrs × 3 exposures/ growing season	660-3140	(0.25-1.20)	No yield effect	Wilhour, et al. 1978
Spring Wheat	I	3 hrs × 7 exposures/ growing season	660-3140	(0.25-1.20)	No yield effect	
Spring Wheat	I	72 hrs/wk/growing season	80- 260	(0.03-0.10)	No yield effect	
Spring Wheat	I	72 hrs/wk/growing season	390	(0.15)	No yield effect	
Durum Wheat	I	3 hrs × 3 exposures/ growing season	660-3140	(0.25-1.20)	No yield effect	Wilhour, et al. 1978
Durum Wheat	I	3 hrs × 7 exposures/ growing season	660-3140	(0.25-1.20)	No yield effect	
Durum Wheat	I	72 hrs/wk/growing season	80- 260	(0.03-0.10)	No yield effect	
Durum Wheat	I	72 hrs/wk/growing season	390	(0.15)	42% yield loss (N.S.)	

TABLE 8-3. (continued)

Plant species	Type of exposure	Time	Concentration $\mu g/m^3$ (ppm)	Effects	References
Barley	I	3 hrs × 3 exposures/ growing season	660-3140 (0.25-1.20)	No yield effect	Wilhour et al. 1978
Barley	I	3 hrs × 7 exposures/ growing season	660-3140 (0.25-1.20)	No yield effect	
Barley	I	72 hrs/wk/growing season	80- 260 (0.03-0.10)	No yield effect	
Barley	I	72 hrs/wk/growing season	390 (0.15)	44% yield loss (N.S.)	
Alfalfa	I	3 hrs × 3 exposures/ growing season	660-3140 (0.25-1.20)	No yield effect	Wilhour et al. 1978
Alfalfa	I	3 hrs × 7 exposures/	660-3140 (0.25-1.20)	No yield effect	
Western Wheat Grass	C	growing season	50-260 (0.02-0.10)[a]	Increasing S content with increasing SO_2	Dodd et al., 1978
Prairie June Grass	C	growing season	50-260 (0.02-0.10)[a]	Significant decrease in digestible protein	

P = Peak (short-term, 8 hrs or less), I = intermittent and C = continuous exposure.

*Statistically significant change at $P < 0.05$.

**Statistically significant change at $P < 0.01$.

N S. Results were not significant at $P < 0.05$ probability level.

[a] Values are for a geometric mean.

TABLE 8-4. THE DEGREE OF INJURY OF EASTERN WHITE PINE OBSERVED AT VARIOUS DISTANCES FROM THE SUDBURY SMELTERS FOR 1953-63

Forest Sampling Station[a] (Distance and Direction from Sudbury)	Trees with Current Year's Foliage Injured in August 1963 (%)	Trees with 1-Year-Old (1962) Foliage Injured		Trees with 2-Year Old Foliage		Net Annual Average Gain or Loss in Total Volume, 1953-1963 (%)	Annual Average Mortality 1953-1963 (%)	Degree of SO_2 Damage	Average SO_2 Concentration[b] for Total Measurement Period 1954-1963 μg/m³ (ppm)
		June 1963 (%)	August 1963 (%)	Injured in June 1963 (%)	Lacking in August 1963 (%)				
West Bay (19 miles NE)	2.0	38.0	77.9	96.0	20.6	-1.3	2.6	Acute and chronic injury	115 (0.045)
Portage Bay (25 miles NE)	1.1	21.5	55.6	77.0	15.2	-0.5	2.5	Mostly chronic and little acute injury	45 (0.017)
Grassy to Emerald Lake (40-43 miles NE)	0.4	2.5	16.7	37.5	9.1	+1.8	1.4	Very little chronic injury	21 (0.008)
Lake Matinenda (93 miles W)	0.6	0.3	2.1	10.1	3.9	+2.1	0.5	Control: no SO_2 injury	3 (0.001)[c] (Sturgeon Falls)
Correlation Coefficient (r)	0.96	0.96	0.93**	0.90**	0.94**	0.90**	0.81		

[a] Linzon (1971) (Pollutants other than SO_2 were not measured and the monitoring was done several miles from the pine stands.)

[b] Dreisinger (1965)

[c] Data for 5-month growing season-1971

*$p < 0.05$

**$p < 0.10$

Derived from Linzon, 1980.

TABLE 8-5. AMBIENT EXPOSURES TO SULFUR DIOXIDE THAT CAUSED INJURY TO VEGETATION[a]

Exposure where injured SO_2 Concentration, µg/m³ (ppm) for averaging periods of:				Plant
1 h	2 h	4 h	8 h	
1070 (0.41)	1000 (0.38)	860 (0.33)	790 (0.30)	Willow
1070 (0.41)	1000 (0.38)	890 (0.34)	680 (0.26)	Larch
1100 (0.42)	1020 (0.39)	680 (0.26)	340 (0.13)	Quaking aspen
1180 (0.45)	890 (0.34)	660 (0.25)	550 (0.21)	Bracken fern
1180 (0.45)	920 (0.35)	660 (0.25)	550 (0.21)	White pine
1210 (0.46)	1000 (0.38)	730 (0.28)	550 (0.21)	White birch
1210 (0.46)	1180 (0.45)	1130 (0.43)	550 (0.21)	Bean
1210 (0.46)	1130 (0.43)	1130 (0.43)	550 (0.21)	Alder
1360 (0.52)	1150 (0.44)	760 (0.29)	520 (0.20)	Jack pine
1470 (0.56)	1020 (0.39)	680 (0.26)	390 (0.15)	Buckwheat
1650 (0.63)	1150 (0.44)	630 (0.24)	310 (0.12)	Barley
1650 (0.63)	1550 (0.59)	890 (0.34)	450 (0.17)	Oats, peas, rhubarb
1680 (0.64)	1470 (0.56)	1130 (0.43)	1000 (0.38)	Lettuce, tomato, potato
1730 (0.66)	1130 (0.43)	970 (0.37)	520 (0.20)	Large-toothed aspen
1730 (0.66)	1180 (0.45)	1150 (0.44)	860 (0.33)	Austrian pine
1730 (0.66)	1410 (0.54)	1050 (0.40)	550 (0.21)	Timothy
1830 (0.70)	1210 (0.46)	710 (0.27)	370 (0.14)	Red clover
1830 (0.74)	1650 (0.63)	1390 (0.53)	1020 (0.39)	Raspberry
1990 (0.76)	1410 (0.54)	760 (0.29)	370 (0.14)	Radish
2040 (0.78)	1810 (0.69)	1150 (0.44)	790 (0.30)	Red pine
2150 (0.82)	1700 (0.65)	1180 (0.45)	680 (0.26)	Balsam poplar
2150 (0.82)	1700 (0.65)	1620 (0.62)	1210 (0.46)	Sugar maple
2280 (0.87)	1940 (0.74)	1440 (0.55)	760 (0.29)	Celery
2280 (0.87)	2070 (0.79)	1830 (0.70)	1310 (0.50)	White spruce
2310 (0.88)	1680 (0.64)	1100 (0.42)	710 (0.27)	Swiss chard
2330 (0.89)	2150 (0.82)	1600 (0.61)	1070 (0.41)	Red oak
2460 (0.94)	2330 (0.89)	1830 (0.70)	1180 (0.45)	Cabbage
2830 (1.08)	2070 (0.79)	1310 (0.50)	660 (0.25)	Carrot, cucumber
2990 (1.14)	1970 (0.75)	1180 (0.45)	600 (0.23)	Witch hazel
3430 (1.31)	2020 (0.77)	1180 (0.45)	600 (0.23)	Beet, turnip
3510 (1.34)	2380 (0.91)	1310 (0.50)	890 (0.34)	Spinach

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[a] Derived from Dreisinger and McGovern, and reprinted from NAS Sulfur Oxides Document, 1978, p. 6-29.

"one could be reasonably certain of avoiding foliar injury to vegetation in the study area if concentrations did not exceed 1830, 1050, 680 or 470 μg/m^3 (0.70, 0.40, 0.26, or 0.18 ppm) as 1-, 2-, 4-, or 8-h means, respectively."

These conclusions appear to be based on the following discussion in the Dreisinger and McGovern (1970) paper:

> "What levels of sulphur dioxide are considered potentially injurious to vegetation? It was shown in previous reports (Dreisinger, 1965, 1967) that most of the injury to vegetation that occurred near recorder stations in the Sudbury area happened when concentrations and durations of sulphur dioxide reached or exceeded the following levels during the day time.
>
> 0.95 ppm for 1 hour)
> or 0.55 ppm for 2 hours)
> or 0.35 ppm for 4 hours) Intensity factor of 100
> or 0.25 ppm for 8 hours)
>
> "It is not true that injury always occurs when these levels are reached or exceeded. Whether or not damage actually happens depends, to a large extent, on the environmental conditions prevailing at the time of the fumigation. There have been occasions in the Sudbury area when levels two, three, and even four times those listed above were recorded and no injury occurred. Those fumigations simply happened at times when vegetation was resistant."
>
> "Conversely, on occasions, some very sensitive species were injured with levels approximately 25% less than the above. In each case the injury occurred during June or July, which, for the Sudbury area, are the months when the most rapid growth of vegetation takes place. Hot, humid weather prevailed in most cases, producing conditions extremely favourable for a high photosynthetic rate and in turn increased susceptibility to sulphur dioxide."
>
> "The occasions, however, when this happened, were quite rare, only nine times in a ten-year period, and the injury that did result was not extensive so that for all intents and purposes, it is only when all SO_2 levels reach the 100-intensity mark or more that the potential for wide-spread injury is present."

The general conclusions of Dreisinger and McGovern (1970) have essentially been reinforced by Jones et al (1974) and McLaughlin (1981).

McLaughlin (1981), although recognizing that, especially in arid regions of the U.S., many species of vegetation would not be visibly injured even in some cases by SO_2 concentrations of 11 ppm for 2 hours as suggested by Hill et al (1974), focused on a subset of vegetational species in the more hot and humid Southeast which he believed to represent a spectrum of plant response ranging from sensitive to tolerant (McLaughlin and Lee, 1974). McLaughlin concluded that 20% of the selected species would show visible injury at peak (5 minutes), 1-, and 3-hour concentrations of 3140, 1700, and 920 μg/m^3 (1.20, 0.65, and 0.35 ppm), respectively.

Jones et al. (1974) also reported, as previously presented in Table 8-2, a wide range of SO_2 concentrations which, at peak (maximum of 5 minutes), 1-hour, and 3-hour durations, injured different species of plants that were classified as sensitive, intermediate, or resistant to SO_2 exposures. However, because of the many randomly occurring environmental factors affecting injury, e.g., plant resistance, temperature, moisture, etc., Jones (1981) recommended that an analysis of probability of injury occurrence should be applied prior to any generalizations regarding the specific concentration of SO_2 causing plant injury.

The work of Hill et al. (1974) also reinforces the widely accepted statement by Dreisinger and McGovern (1970) that hot, humid weather produces conditions extremely favorable for increased plant susceptibility to sulfur dioxide.

The studies of Vins and Mrkva (1973) and Materna et al. (1969), although reporting foliar and growth increment losses in forest trees as being due to SO_2, were conducted in areas of fluoride contamination or used only sporadic monitoring schedules, respectively. Pollution gradients were evident and SO_2 exposure was probably involved, but conclusive proof of losses attributed to SO_2 was not presented as was done for the Sudbury area.

Table 8A-3 (Appendix 8A) summarizes the results of tree studies that have utilized artificial exposure chamber systems under laboratory conditions. Two of the laboratory studies listed used levels close to ambient concentrations. Houston (1974), for example, reported significant effects on pine seedlings at levels as low as 390 $\mu g/m^3$ (0.15 ppm) SO_2 for 6 hr; however, the use of selected clones known to be sensitive to SO_2 hinders extrapolation of these findings to ambient conditions. Most of the remainder of the studies presented in Table 8A-3 have used doses above expected occasional exposures under field conditions.

The effects of exposing seedling tree species to SO_2 levels within or approaching the ambient range are summarized in Table 8-6. Concentrations of greater than 660 $\mu g/m^3$ (0.25 ppm) SO_2 for 2 hours were required to induce slight injury to several pine species (Berry, 1971), but overall trends for increasing foliar injury do not follow increasing dose for conifers per se. Smith and Davis (1978) exposed several conifers (pine, spruce, fir and Douglas fir) to 2620 $\mu g/m^3$ (1.0 ppm) SO_2 for 4 hours or 5240 $\mu g/m^3$ (2.0 ppm) SO_2 for 2 hours and only the pines developed necrotic tips at the higher concentration. However, studies done with tree seedlings under artificial conditions are difficult to extrapolate to expected yield-loss effects under field conditions. Of the studies reviewed and accepted without caveats, none demonstrated significant height or annual increment growth effects.

In an evaluation of SO_2 effects on photosynthesis, Keller (1977) used field chambers to expose potted white fir, Norway spruce and Scotch pine to three concentrations of SO_2 (130, 260 and 520 $\mu g/m^3$; 0.05, 0.10, and 0.20 ppm) and a control of 0.0 $\mu g/m^3$ for 10 weeks each during the spring, summer, and fall, respectively. Several types of photosynthetic response rates were obtained; however, trends of decreasing rate occurred as concentrations increased, especially when administered during the 10-week spring period. Effects were less during the summer and fall periods and spruce responded positively to SO_2 concentrations of 130 $\mu g/m^3$ (0.05 ppm) during the initial part of the fall period. Keller (1977) also reported that even with the most severe depression of photosynthesis, there were no visible foliar symptoms in evidence. Keller (1980) utilized a similar fumigation system and similar SO_2 concentrations to study effects on the annual ring width in two clones of Norway spruce. He reported significantly depressed CO_2 uptake with higher levels (260 and 520 $\mu g/m^3$; 0.10 and 0.20 ppm SO_2, 10

TABLE 8-6. SUMMARY OF THE EFFECTS RESULTING FROM THE EXPOSURE OF SEEDLING TREE SPECIES IN THE LABORATORY

Plant species	Type of exposure	Time	Concentration µg/m³ (ppm)	Effects	References
Scotch Pine	P	1 hr-6 hrs	1310-5240 (0.5-2.00)	Foliar injury - except at 5 hrs at 0.5 ppm	Smith and Davis, 1978
Eastern White Pine	P	2 hrs - 6 hrs	6.6-1310 (0.025-0.5)	Foliar injury even on tolerant clones at 6 hrs at 0.15 ppm	Houston, 1974
Red Pine	P	2 hrs	660 (0.25)		Berry, 1972
Loblolly Pine	P	2 hrs	660 (0.25)		
Shortleaf Pine	P	2 hrs	660 (0.25)	Foliar injury; most sensitive period of plant 8-10 wks	Berry, 1972
Virginia Pine	P	2 hrs	660 (0.25)		
Slash Pine	P	2 hrs	660 (0.25)		
Jack Pine	P	2 hrs	660 (0.25)	Foliar injury	Berry, 1974
Jack Pine	C	24 hrs	470-520 (0.18-0.2)	Inhibition of foliar lipid synthesis; inhibition reversable	Malhotra and Kahn, 1978
Balsam Fir	P	2, 4, 5 hrs	5240, 2620 and 1310 (2.0, 1.0 and 0.5)	Foliar injury only at 4 hrs, then less than 4%	
Fraser Fir	P	2, 4, 5 hrs	5240, 2620 and 1310 (2.0, 1.0 and 0.5)	Foliar injury only at 4hrs, then less than 4%	
White Fir	P	2, 4, 5 hrs	5240, 2620 and 1310 (2.0, 1.0 and 0.5)	Foliar injury only at 4 hrs, then less than 4%	
Blue Spruce	P	2, 4, 5 hrs	5240, 2620 and 1310 (2.0, 1.0 and 0.5)	Foliar injury only at 4 hrs, then less than 4%	Smith and Davis, 1978
Douglas Fir	P	2, 4, 5 hrs	5240, 2620 and 1310 (2.0, 1.0 and 0.5)	Foliar injury only at 4 hrs, then less than 4%	
Austrian Pine	P	2, 4, 5 hrs	5240, 2620 and 1310 (2.0, 1.0 and 0.5)	Foliar injury except at 4 hrs and 0.5 ppm	
Ponderosa Pine	P	2, 4, 5 hrs	5240, 2620 and 1310 (2.0, 1.0 and 0.5)	Foliar injury except at 5 hrs and 0.5 ppm	
Ponderosa Pine	P	9 hrs/days × 8 wks	1180 (0.45)	Tip necrosis	Evans and Miller, 1975
American Elm	P	6 hrs	5240 (2.0)	Severe foliar injury, defoliation	Constantinidou and Kozlowski, 1979a
American Elm	P	8 hrs	2620 (1.0)	Inhibited stomatal closing	Noland and Kozlowski, 1979
American Elm	C	12 hrs	5240 (2.0)	Stomatal closing induced, increases content when plants fumigated in light	Temple, 1972

TABLE 8-6. (continued)

Plant species	Type of exposure	Time	Concentration μg/m³ (ppm)	Effects	References
Chinese Elm	P	6 hrs	5240 (2.0)	100% leaf necrosis	Temple, 1972
Chinese Elm	C	24 hrs/day for 30 days	1310 (0.5)	7 days for chlorosis	Temple, 1972
Gingko	P	6 hrs	7900 (3.0)	50% leaf necrosis	Temple, 1972
Gingko	C	24 hrs/day for 30 days	1310 (0.5)	14 days for chlorosis	Temple, 1972
Trembling Aspen	P	3 hrs	920; 1700 (0.35; 0.65)	Foliar injury; 2% and 2.3%	Karnosky, 1976
Sugar Maple	C	24 hrs/day for 1 wk	1310 (0.5)	54% reduction in photosynthesis	
Sugar Maple	I	4 hrs/day for 3 wk	1310 (0.5)	43% reduction in photosynthesis	
Black Oak	C	24 hrs/day for 1 wk	1310 (0.5)	48% reduction in photosynthesis	
Black Oak	I	4 hrs/day for 3 wk	1310 (0.5)	74% reduction in photosynthesis	Carlson, 1979
White Ash	C	24 hrs/day for 1 wk	1310 (0.50)	20% reduction in photosynthesis	
White Ash	I	4 hrs/day for 3 wk	1310 (0.50)	7% reduction in photosynthesis	
Norway Maple	C	24 hrs/day for 30 days	1310 (0.50)	12 days for chlorosis	Temple, 1972
Pin Oak	C	24 hrs/day for 30 days	1310 (0.50)	30 days for chlorosis	Temple, 1972
Norway Spruce	C	10 wks	1310 (0.05)	No foliar injury, 25% < volume	Keller, 1980
Norway Spruce	C	10 wks	260 (0.10)	Foliar injury 38% < volume	Keller, 1980
Norway Spruce	C	10 wks	520 (0.20)	Foliar injury 53% < volume	Keller, 1980
Beech Spruce	C	16 wks	130 (0.05)	A few buds killed	Keller, 1978
Beech Spruce	C	in winter	260 (0.10)	Many buds killed	Keller, 1978
Beech Spruce	C		520 (0.20)	50% buds killed	Keller, 1978

[a]Data for this table come from Table 8A-3 in Appendix 8A.

P, C, and I indicate peak (short-term average for 8 hrs or less), continuous and intermittent exposures.

weeks), but a trend was only noted for the 0.05 ppm SO_2 treatment; visible symptoms and trends of reduced cambial growth occurred only at the higher concentration.

Based on the limited data available concerning the nonwoody components of native ecosystems, there appear to be no adverse yield effects below 160 $\mu g/m^3$ (0.06 ppm) SO_2 for 6 weeks (Table 8A-4, Appendix 8A). Most of the acceptable literature deals with long-term exposures (several weeks) and results in SO_2 levels well above ambient atmosphere concentrations. There is some indication of beneficial yield effects below 160 $\mu g/m^3$ (0.06 ppm) SO_2 on one species (i.e., Indian ricegrass). As stated in Section 8.2.8, low doses of SO_2 may increase the productivity of certain crops. Concentrations of 80 to 160 $\mu g/m^3$ (0.03 to 0.06 ppm) continuously for 6 weeks of exposure increased the productivity of desert grass by 8 percent over the control plants which were grown in indigenous soils (Ferenbaugh, 1978). Other studies have also demonstrated beneficial effects, but experimental conditions included a sulfur-deficient growing medium.

More definitive dose-response studies, both with and without the addition of other pollutants (see Section 8.4), are needed before the biologically significant features of typical regional exposure regimes can be positively delineated. The National Crop Loss Assessment Network (NCLAN), which has only recently begun its studies, is a step in this direction.

8.4 EFFECTS OF MIXTURES OF SULFUR DIOXIDE AND OTHER POLLUTANTS

Ambient atmospheres usually contain more than one pollutant. Atmospheric monitoring of long-distance transport of photochemical oxidants and oxidant precursors (Husar et al., 1978; U.S. EPA, 1971), and the presence of acidic precipitation over large areas of the eastern United States (Cogbill, 1976) have documented the fact that emissions from specific sources are mixed with ambient concentrations of one or more pollutants. Extrapolation from results of single-pollutant effects on vegetation under ambient field conditions must be approached with caution because reactions to pollutant combinations under controlled conditions, the interaction of constantly changing environmental factors, and fluctuating pollutant exposures must also be evaluated before a conclusive statement of the importance of such interactions can be made. Reinert (1975) and Reinert et al. (1975) have prepared the most recent reviews of this area of investigation. Some examples of the available literature follow.

8.4.1 Sulfur Dioxide and Ozone

Table 8A-5 (Appendix 8A) summarizes studies on the effects of sulfur dioxide in combination with ozone.

A more-than-additive effect on vegetation was first reported for ozone and SO_2 (Menser and Heggestad, 1966). Tobacco was severely injured by 80 $\mu g/m^3$ (0.03 ppm) ozone and 630 $\mu g/m^3$ (0.24 ppm) SO_2 when the pollutants were combined for either 2 or 4 hours, whereas when used alone neither pollutant produced foliar symptoms.

Since that first report, the effects of mixtures of ozone and SO_2 have been studied using a variety of plant species. Radish and alfalfa plants showed more-than-additive foliar injury after a 4-hr exposure to a mixture of 200 $\mu g/m^3$ (0.10 ppm) O_3 + 260 $\mu g/m^3$ (0.10 ppm) SO_2

(Tingey et al., 1973a), but less-than-additive growth reduction (top and root weights) from an 8 hr/day, 5 day/wk, 5-week exposure of radish (alfalfa total exposure time unknown) to a mixture of 100 $\mu g/m^3$ (0.05 ppm) O_3 + 130 $\mu g/m^3$ (0.05 ppm) SO_2 (Tingey et al., 1971a; Tingey and Reinert, 1975). Greater-than-additive foliar injury effects have also been reported for broccoli and tobacco, while additive or less-than-additive effects have been noted for cabbage, tomato, lima bean, bromegrass, spinach, onion, and soybean (Tingey et al., 1973a). Soybean has exhibited insignificant less-than-additive foliar injury effects (Tingey et al., 1973a) while exhibiting significantly greater-than-additive growth effects (Tingey et al., 1973b).

Most researchers examining the effects of pollutant mixtures have utilized standard means comparisons to express the responses. These tests usually do not adequately evaluate the interaction: the failure of one pollutant to be consistent at different concentrations of the second pollutant. Reinert and Nelson (1980) used sums of squares partitioning by factorial analysis to examine the effects of 1310 $\mu g/m^3$ (0.5 ppm) SO_2 and 490 $\mu g/m^3$ (0.25 ppm) O_3 (4-hr exposures, 4 times, 6 days apart) on Begonia. A significantly less-than-additive effect was found for flower weight in 1 of 5 cultivars. The same technique was utilized by Reinert and Weber (1980) to evaluate the effects of 490 $\mu g/m^3$ (0.25 ppm) O_3 and 660 $\mu g/m^3$ (0.25 ppm) SO_2 (exposed 4 h/day, 3 days/wk, for 11 wks) on soybean (Dare); an additive effect of the pollutant mixture was demonstrated.

Field-grown soybeans (cv Dare) exposed to 200 $\mu g/m^3$ (0.10 ppm) O_3 alone or 200 $\mu g/m^3$ (0.10 ppm) O_3 + 260 $\mu g/m^3$ (0.10 ppm) SO_2 for 6 hr/day for 133 days in field chambers exhibited injury and defoliation. Due to the mixture, injury and yield were increased (9 percent) and decreased (19 percent), respectively, when compared to the ozone-alone treatment, but the differences were not significant (Heagle et al., 1974). Two cultivars of bean exposed to sulfur dioxide and ozone showed interactive effects between these two gases, but the magnitude and direction of the effects depended on the cultivar and on the pollutant concentrations (Jacobson and Colavito, 1976).

Many studies have been conducted on the effects of mixtures of sulfur dioxide and ozone on eastern white pine (Pinus strobus L.) (Costonis, 1973; Dochinger and Heck, 1969; Houston, 1974; Houston and Stairs, 1973). Genetic control of sulfur dioxide and ozone tolerance in this species has been demonstrated for low concentrations of SO_2 (66 $\mu g/m^3$; 0.025 ppm) and O_2 (100 $\mu g/m^3$; 0.05 ppm) for only 6 hr with consistent injury to the exposed sensitive clones (Houston and Stairs, 1973). Houston (1974) later used mixtures of sulfur dioxide and ozone at concentrations simulating actual field conditions and reported that even the lowest concentrations of O_3 (100 $\mu g/m^3$; 0.05 ppm) and SO_2 (130 $\mu g/m^3$; 0.05 ppm) for 6 hr in mixture caused more serious damage than that resulting from either pollutant alone at similar concentrations. A less-than-additive effect on foliar injury was noted when Scotch pine trees were exposed to 660 $\mu g/m^3$ (0.25 ppm) SO_2 and/or 270 $\mu g/m^3$ (0.14 ppm) or 570 $\mu g/m^3$ (0.29 ppm) O_3, 6 hr/day for varying time periods (Neilson et al., 1977).

Fumigation of aspen clones to 100 μg/m^3 (0.05 ppm) O_3 and/or 520 μg/m^3 (0.20 ppm) SO_2 for 3 hr resulted in a number of plants in the mix exhibiting more-than-additive foliar injury (Karnosky, 1976).

Oshima (1978), using enclosed, round chambers in which the air was constantly stirred, conducted a field experiment to assess the effect of SO_2 on the yield of red kidney beans, Phaseolus vulgaris, grown in pots. The beans were exposed to 260 μg/m^3 (0.10 ppm) of SO_2 for 6 hr/day for a total of 335 hours and at the same time were exposed to a gradient of ozone doses. The ozone concentrations were those that resulted when ambient air at Riverside, California, was passed directly into the chamber or when 25, 50, 75 and 100 percent of the ambient air was passed through a charcoal filter. The temperature, light, and humidity approximated that of ambient air. Plants were in the chambers for a total of 78 days. An interaction between ozone and SO_2 was documented in the 50 percent carbon filtered treatment (i.e., 50 percent ambient air) (51 ppm-hrs) and produced a significant reduction in yield (37 percent) and plant biomass. The data also suggested the possibility of an interaction in the 25 percent ambient (75 percent filtered treatment) (28.22 ppm-hrs) in that yield was reduced 17 percent, but this reduction was not significant ($p = 0.20$). Exposure of beans to SO_2 alone at 260 μg/m^3 (0.10 ppm) did not produce detectable plant or yield responses nor was an interactive effect observed when ozone doses exceeded 51 ppm-hrs; however, ozone alone at the same concentrations did significantly ($\geq$ 65%) reduce yield.

8.4.2 Sulfur Dioxide and Nitrogen Dioxide

The occurrence of SO_2 and nitrogen dioxide (NO_2) together in the atmosphere has been associated with power plant plumes as well as mobile sources. However, ambient concentrations of NO_2 seldom reach the injury threshold, and the literature for NO_2 suggests that any injury associated with NO_2 results from interactions with other pollutants (Jacobson and Hill, 1970).

No injury occurred to oats, beans, soybeans, radish, tomato, or tobacco following exposure for 4 hr with up to 3760 μg/m^3 (2 ppm) NO_2 or 1310 μg/m^3 (0.50 ppm) SO_2. However, following exposures at 0.10 ppm of each gas for 4 hr, injury was noted on all species; at 0.05 ppm of each gas, slight injury was noted on all species except tomato (Tingey et al., 1971a). A greater-than-additive suppression of the apparent photosynthetic rate of alfalfa was obvious when treated with to 660 μg/m^3 (0.25 ppm) of SO_2 and/or 470 μg/m^3 NO_2 for 2 hr (White et al., 1974). At 0.15 ppm of each gas singly, there were no measurable effects, but a 7-percent suppression of apparent photosynthetic rates was noted in the mixture (White et al., 1974).

Field exposure of seven different species of plants indigenous to the cold desert areas of the southwestern United States to 1310-28820 μg/m^3 (0.50-11.0 ppm) SO_2 singly, or 1310-28820 μg/m^3 (0.50-11.0 ppm) SO_2 and 190-9400 μg/m^3 (0.10-5.00 ppm) NO_2 combined in 2-hr fumigations resulted in no evidence of more-than-additive foliar injury (Hill et al., 1974).

More-than-additive foliar injury was noted on radish leaves exposed for 1 hr to 1310 μg/m^3 (0.50 ppm) SO_2 and/or 940 μg/m^3 (0.50 ppm) NO_2. No interactive effects were found for other plants tested (oats, Swiss chard, and pea) (Bennett et al., 1975). More-than-additive effects have been associated with the enzyme activity in pea plants exposed to 520 μg/m^3 (0.20 ppm) SO_2 and 190 μg/m^3 (0.10 ppm) NO_2 for 6 days; peroxidase activity was increased and ribulose-1,5 diphosphate carboxylase activity was decreased (Horsman and Wellburn, 1975). Some selected SO_2 + NO_2 combination studies are shown in Table 8A-6 (Appendix 8A).

8.4.3 Sulfur Dioxide and Hydrogen Fluoride

Suppression of linear growth and leaf area in the absence of foliar injury of Koethen orange plants exposed to SO_2 (2100 μg/m^3; 0.80 ppm) and/or hydrogen fluoride (2.3-13.0 ppb) for 23 days was no greater than additive. Satsuma mandarin plants exposed to the same conditions for 15 days exhibited only additive foliar injury and no growth suppression at all (Matsushima and Brewer, 1972). Greater-than-additive foliar injury was exhibited by barley and sweet corn from to 160-210 μg/m^3 (0.06-0.08 ppm) SO_2 and/or 0.60-0.90 ppb hydrogen fluoride for 27 days. Using higher concentrations of SO_2 for only 7 days resulted in simply additive foliar injury. Pinto beans were not injured in any of the treatments (Mandl et al., 1975).

8.4.4 Sulfur Dioxide, Nitrogen Dioxide and Ozone

Fujiwara et al. (1973) combined SO_2, NO_2, and O_3 at concentrations ranging from 0 to 0.2 ppm in an artificially controlled environment and exposed peas and spinach for 5 hr. Ozone was the most injurious, SO_2 was next, and NO_2 elicited only minor injury. More-than-additive foliar injury followed treatment with SO_2 + O_3, but only additive effects were observed with SO_2 + NO_2 or NO_2 + O_3. The addition of NO_2 to the O_3 + SO_2 had little effect on foliar injury. Reinert and Gray (1981) examined the effects of 0.2 or 0.4 ppm each of O_3, SO_2, and NO_2 (3- or 6-hr fumigations) on the growth of radish. The main effect of each pollutant and the potential interactive effects of each mixture were examined through partitioning. Sulfur dioxide depressed the root/shoot ratio at both 520 and 1050 μg/m^3 (0.2 and 0.4 ppm); however, when NO_2 and SO_2 were both present there was a greater-than-additive depression of the root/shoot ratio at 1050 μg/m^3 (0.4 ppm).

8.4.5 Summary

As can be seen from the preceding analysis of research dealing with pollutant mixtures, plant species vary in their response and the type of response (additive, less-than-additive, greater-than-additive) may depend on the parameter measured. Understanding of how pollutant combinations influence plant growth and development, and how environmental factors can modify those responses is still fragmentary. Insufficient information exists to determine the influence of pollutant sequencing during combination exposures, meteorological influences, the effect of various cultural practices, and many other variables in relation to vegetational effects induced by SO_2 combined with other pollutants.

There is a need to determine the best technique for evaluating the effects of pollutant mixtures. Only recently has the partitioning technique been utilized to elucidate the effects of individual pollutants in pollutant mixtures. This technique allows for separation of the main effects of each pollutant and also provides a statistical test of the significance of potential interactions between pollutants.

Analysis of the data indicates that interactions can occur between pollutants and, because the occurrence of pollutant mixtures is typical of the ambient atmosphere, knowledge of interactive effects is very important. However, the nature of the effects of pollutant mixtures is extremely complex. Most research studies have necessarily taken a rather simplistic approach to this complex problem. It is, therefore, difficult to relate these relatively few results to "real world" situations.

8.5 EFFECTS OF NON-POLLUTANT ENVIRONMENTAL FACTORS ON SULFUR DIOXIDE PLANT EFFECTS

The physical environment plays an extremely important role in determining response to SO_2. Most of the information accumulated at present deals with the factors leading to or inhibiting the ingress of SO_2 into stomata and the immediate plant response as determined by the metabolism of the plant at the time of exposure. The metabolic state of the plant is likewise affected by the physical environment. As discussed in the following sections, the response of any given plant species may also be quite different from any other given species grown under identical physical conditions.

8.5.1 Temperature

Temperature plays an important part not only in determining the metabolic rate of the plant, but also in determining (with moisture, fertility, and light) species diversity and richness of a given ecosystem (NAS, 1978). The primary path of SO_2 entry into the leaf is through stomata. Temperature exerts an effect on the guard cells that control the stomatal opening and closing and thus the entry of SO_2. Temperature regimes that increase the physiological activity of the plant may also increase the plant response to SO_2 (Heck and Dunning, 1978). It is generally believed that plant sensitivity increases with temperature over a wide range, from about 4^0 to 35^oC (Guderian, 1977; Rist and Davis, 1979). Several studies suggest that the greater resistance of conifers to SO_2 in the winter is attributable to lower rates of physiological activity (NAS, 1978). However, according to Guderian and Stratmann (1968), in areas with SO_2 emissions, winter wheat and winter rye are more severely injured than the summer varieties. Guderian (1966) interpreted this to be due to gas exchange taking place through stomata at temperatures as low as -2^oC.

8.5.2 Relative Humidity

Relative humidity exerts an important control over plant sensitivity to SO_2 by affecting stomatal opening and closing (Bonte et al., 1975; Majernik and Mansfield, 1970; Mansfield and Majernik, 1970; Buron and Cornic, 1973) and by affecting the internal leaf resistance to SO_2 flux (McLaughlin and Taylor, 1981). Although plant sensitivity increases with relative humidity, Setterstrom and Zimmerman (1939) found rather large changes ($\geq$ 20 percent) were required to cause a change in plant sensitivity once RH levels become $\geq$ 40 percent. McLaughlin and Taylor (1981), in laboratory studies, found 2-to 3-fold increases in SO_2 uptake by kidney beans over a range of SO_2 concentrations (420-1680 μg/m^3; 0.16-0.64 ppm) as relative humidity was increased from 35 percent to 78 percent during exposure times of ≤ 3 hours. According to Zimmerman and Crocker (1934), although relative humidity is important in governing sensitivity and, consequently, the sensitive plant population, it is not as important as the tissue turgidity which may be influenced by soil moisture as well as relative humidity. Based on the water relations in certain trees and their sensitivity at relative humidities of over 75 percent, 50 to 75 percent, and below 50 percent, Halbwachs (1976) has rated plants as sensitive, intermediate, and tolerant, respectively.

8.5.3 Light

Light controls stomatal opening and thus plant sensitivity. Plants are more tolerant when fumigated in darkness with SO_2 or when held in the dark for several hours before exposure (Zimmerman and Crocker, 1934). This relationship is complex, since injury is greater if the night exposure follows a daylight exposure (NAS, 1978).

Setterstrom and Zimmerman (1939) observed that buckwheat grown at a light intensity of 35 percent or less of full sunlight was more sensitive to SO_2 than when grown under full sunlight. Other investigators have found that injury was more severe when tomato stems and foliage were fumigated on clear days than on cloudy days (NAS, 1978).

Plants seem to be more sensitive from midmorning to midafternoon, despite the high light intensity that might continue after midafternoon (Rennie and Halstead, 1977; Thomas and Hendricks, 1956). At the same time, plants may be more sensitive in the morning during good weather, but may become more sensitive if temperature and light increase in late afternoon in bad weather (Van Haut and Stratmann, 1970).

8.5.4 Edaphic Factors

Soil factors influence directly and indirectly the responses of plants to SO_2. Soil fertility, moisture, and soil physics directly influence plant sensitivity to SO_2 (NAS 1978). Adequate soil moisture and the resultant stomatal opening have been shown to increase the degree of plant sensitivity, whereas wilting conditions confer tolerance (Setterstrom and Zimmerman, 1939; Zahn, 1963a; Zimmerman and Crocker, 1934). As long as plants are grown with an inadequate supply of water, they are less sensitive to SO_2 than are plants grown with an adequate supply, even though the moisture content of the soil is the same at the time of fumigation (Setterstrom and Zimmerman, 1939). Withholding water from some crops during periods of high pollution risk has been suggested as a technique to reduce injury (Brandt and Heck, 1968).

Soil fertility has a significant influence on plant response to SO_2. Some plants become more tolerant to SO_2 upon fertilization (Enderlein and Kastner, 1976; Zahn, 1963b). However, with eastern white pine, increased nitrogen, phosphorus, and potassium concentrations in the greenhouse raised tolerance (decreased needle necrosis) in sensitive clones but did not prevent chlorotic banding in the field (Cotrufo and Berry, 1970). Nitrogen and sulfur deficiencies were correlated with increased tolerance to SO_2 in tobacco and tomato (Leone and Brennan, 1972). Conversely, nutrient deficiencies increased SO_2 sensitivity in alfalfa (Setterstrom and Zimmerman, 1939). Fertilization of several dicotyledons with a complete fertilizer has been effective in decreasing their sensitivity to SO_2, but similar treatment of monocotyledons like oats and barley has been ineffective (Van Haut and Stratmann, 1970; Zahn, 1963b).

8.5.5 Sulfur Dioxide and Biotic Plant Pathogen Interactions

Plant disease is caused by the interaction of a plant and a pathogen acting under suitable environmental conditions. The influence of SO_2 directly or indirectly on the interrelationships of a given plant and its possible biotic pathogens has been difficult to investigate. Additionally, whenever the variables of the physical environment are considered within such experimental sequences, the subject becomes even more difficult to examine. Heagle (1973) and Laurence (1978) have provided the most recent reviews of the interaction between air pollutants and plant parasites.

In seven of nine plant diseases reported in SO_2-related studies as reviewed by Laurence (1978), there was either no effect or a reduction in disease development demonstrated. Disease increased only in needle cast of pine (Chiba and Tanaka, 1968), and increased virus titer of southern bean mosaic virus has been reported by Laurence et al. (1981).

In a recent study by Laurence (1979) maize and wheat were exposed to 0.10 or 0.15 ppm SO_2 for either 2 or 10 days and innoculated at various times with Helminthosporium maydis or Puccinia graminis. The ability of these fungi to infect either corn or wheat, respectively, was inhibited by SO_2 exposures; greater inhibition occurred if plants were fumigated prior to inoculation attempts.

Studies done under ambient conditions without monitoring of other pollutants have suggested a decrease in disease incidence in areas of higher SO_2 pollution with the possible exception of those pathogens that are better able to invade weakened plants. If SO_2 exposure has resulted in an overall weakened condition, other agents, such as root-invading fungi, may be able to gain entrance into an otherwise resistant host. Such is the suspected reason for increased incidence and severity of attack by the root pathogen Armillaria mellea in trees weakened by SO_2 (Donaubauer, 1968; Jancarik, 1961; Kudela and Novakova, 1962).

The effects of SO_2 on infection by organisms other than fungi have also been studied. Abies concolor (white fir) and A. veitchii were severely attacked by plant lice in an environment of high SO_2, but Pinus strobus (white pine) was attacked less and P. Griffithii (Himalayan pine) and P. sylvestris (Scotch pine) were not attacked (Stewart et al., 1973).

The direct influence of SO_2 on plant pathogenic fungi has been demonstrated and a review published by Saunders (1973); no direct effects of SO_2 on plant pathogenic bacteria have been reported.

8.6 PLANT EXPOSURE TO PARTICULATE MATTER

8.6.1 Deposition Rates

Deposition of particles is strongly dependent on particle size. Most sulfates and nitrates are found in the size range of 0.1 to 1.0 µm, and very little information is available on the deposition rate for these particles (see Chapter 6). Shinn (1978) divided particulate deposition into three categories based on particle size:

CATEGORY 1. Particles more than 10 μm in diameter; includes dust and spores.

CATEGORY 2. Particles between 1 μm and 10 μm in diameter where the collection efficiency is highly dependent on the particle diameter.

CATEGORY 3. Submicrometer particles between 0.1 and 1.0 μm in diameter, which have a nearly constant collection efficiency.

Current experimental data suggest that collection efficiencies in Category 3 are less than one-tenth of those in Category 2 (Shinn, 1978). According to Clough (1975), in the range of wind speeds normally encountered, the larger particles in the atmosphere are much more efficiently collected than the smaller fraction.

Little (1977) evaluated the effects of leaf surface texture on the deposition of monodisperse polystyrene aerosols on the leaf surfaces. Rough and hairy leaf discs collected 5.0 μm particles up to seven times more efficiently than did smooth leaves. Very large differences in particle deposition velocities were observed between the laminas, petioles, and stems of each species. The velocity of deposition of particles to plant surfaces varies according to both wind speed and particle size.

Further information on atmospheric transport, transformation and deposition of particulate matter may be found in Chapter 6 of this document.

8.6.2 Routes and Methods of Entry Into Plants

8.6.2.1 Direct Entry Through Foliage--Foliage is continuously subjected to natural and man-made coarse particles that are insoluble or sparingly soluble in water. Coarse particles in general are too large to enter leaves through stomata. Based upon a review by Meidner and Mansfield (1968) that presented stomatal data for 27 species of plants (e.g., pine, oak, corn, soybean, and tobacco) the overall average pore (opening) width was 6 micrometers and accounted for 0.15 to 2.0 percent of the total average stomatal area. In certain cases, such as with cement kiln dusts (Lerman and Darley, 1975) and other types of aggregate particles (Smith, 1973), a limited amount of stomatal clogging can occur. This is apparently dependent on the statistical probability of the particulate matter falling on the stomata, the size of the particle, and the stomatal aperture. In many plants, the stomatal opening is on the lower surface. Cement kiln dust forms a crust on leaves, twigs, and flowers. According to Czaja (1961), crusts of this type form because some portion of the settling dust consists of calcium aluminosilicates typical of the clinker from which cement is made. Hydration of the dust on the leaf surface results in the formation of a gelatinous calcium aluminosilicate hydrate, which later crystallizes and solidifies to a hard crust.

When coarse particles are water soluble or have some water-soluble components, plant uptake of ions from the leaf surface does occur. Because of analytical difficulties, the exact magnitude of the uptake is difficult to measure. Since it is not possible to predict the efficiency of any washing procedure used to remove particles from the leaf surface, it is difficult to delineate and separate the concentration of a given element on the leaf surface from its concentration inside the tissue. In addition, leaching of elements from within the tissue is known to occur during the washing (Little, 1973).

Smith (1973) evaluated metal contamination of urban woody plants by using a variety of washing procedures. Indirect evidence from all these studies suggests that a variable concentration of metals originating from coarse particles can accumulate in plant foliar tissue through direct uptake.

At this time, one of the significant problems in deriving conclusions concerning the magnitude of direct foliar deposition and uptake of atmospheric particulates is the lack of coordinated data on size and frequency distribution of the particles and their chemistry, rates of deposition, and dose in conjunction with changes in tissue concentrations over time relative to background conditions.

8.6.2.2 Indirect Entry Through Roots--Many of the inorganic constituents of particulate air pollutants occur naturally in the soil; others of equal or greater importance do not. Deposition of these pollutants may increase the soil concentrations of the chemical species in question. Some of the chronic effects caused by particulate air pollutants may result from changes in soil physics and chemistry and from increased plant uptake of either the added elements associated with the particles themselves or some other soil-borne elements made more available by the influence of the deposited particles (Guderian, 1980).

It should be recognized that only a portion of the total elemental content of the soil is available at a given time for plant absorption (Brady, 1974). As uptake of elements proceeds, there may be a redistribution of nutrients or toxicants in the soil.

The availability of nutrients or other chemical elements from the soil is strongly influenced by type, chemical composition, and acidity of the soils. Plant nutrients, when present in optimal amounts, are usually available at a neutral pH; however, when the soil becomes acidic, toxic elements such as aluminum become available.

8.7 REACTION OF PLANTS TO PARTICLE EXPOSURE

8.7.1 Symptomatology of Particle-Induced Injury

Particle-induced injury to plants has most often been associated with sustained accumulation of particles such as dust or fly ash. Few investigations have dealt with direct or indirect chemical interactions at the plant surface or subsequent effects. The toxicity of accumulated heavy metals in soils has been established for several plant species.

The various forms of particles and their associated impacts on plants have been reviewed (Darley, 1966; Lerman and Darley, 1975; U.S. Department of Health, Education, and Welfare, 1969; U.S. EPA, 1977). Krupa et al. (1976) and Linzon (1973) have also prepared extensive reviews of heavy metal deposition and impact. Tolerance of plants for heavy metals and fine particles and their bioenvironmental impacts have also been reviewed (U.S. EPA, 1975).

Dusts--Dust directly affects plants by coating exposed plant parts, including leaves, stems, flowers, and fruits (Jennings, 1934; Katz, 1967; Linzon, 1973). Depending on the chemical nature of the particles and environmental conditions, deposits may accumulate as dry dusts, as encrustations in the presence of free moisture, or as greasy films or tars. Encrustations of dust particles on leaves result in reduced gas exchange, increased temperature, reduced photosynthesis, and eventual yellowing and tissue desiccation (Daessler et al., 1972; Parish, 1910).

Terminal growth reduction and chlorosis of 2-year needles of hemlocks coated with heavy limestone dust deposits have been reported. Manning (1971) also reported that fungal propagules increased, but bacterial numbers decreased on such needles. Brandt and Rhoades (1972) reported long-term changes in plant community structure and species composition, and later indicated that radial growth rates were reduced in the tree species involved (Brandt and Rhoades, 1973). On the exposed site they demonstrated a reduction in radial growth of red maple (18 percent, chestnut oak (29 percent), and red oak (23 percent) but a 76-percent increase in radial growth of tulip poplars as compared with representatives of these species growing on a similar but nonexposed site. Reduction in growth of the dominant species (oak, maple) probably gave a competitive advantage to tulip poplar and a greater than expected increase may have occurred.

The primary result of the deposition of limestone dusts has been the raising of pH levels of the environment, particularly tree bark. The substrate pH changes were followed by lichen community changes in which replacement of acid-tolerant lichen communities by more alkaline-tolerant species occurred (Gilbert, 1976). A recovery of lichen diversity resulted in areas where SO_2 pollution was of importance prior to limestone dust emission. No exact pollutant dose of either limestone dusts or SO_2 was reported. Winter SO_2 levels were estimated to average 65 $\mu g/m^3$.

Cement kiln dusts collected from precipitators have been applied to vegetation. Visible effects were demonstrated on beans following application of particles > 10 µm at rates 0.06 $mg/cm^2/day$ to 0.38 $mg/cm^2/day$ for 2-3 days. The lower dose induced a slight reduction in carbon dioxide exchange, and the two higher doses reduced carbon dioxide uptake by 16-32 percent (Darley, 1966).

The accumulation of dust caused increased reflection of solar radiation at wavelengths of 400 to 750 nm and has been demonstrated to reduce photosynthesis (Ricks and Williams, 1974). Conversely, increased absorption of solar radiation by dusted leaves at wavelengths 750-1350 nm has been demonstrated to lead to heat stresses within the leaf tissues (Spinka, 1971).

Growth and yield effects induced by the accumulation of dust have recently been reviewed (U.S. EPA, 1977). Conflicting reports of yield increases and decreases from such accumulation appear to be caused by variations in doses applied, substrate nutrient balances and pH, and other specific physiological interferences with processes such as pollination of fruit trees (Anderson, 1914).

Dusts, therefore, have only been considered of importance to vegetation growing near emission sources. Accumulation of dusts has been demonstrated to reduce photosynthesis and radial-increment growth of some forest tree species but has increased them in other species.

The phytotoxicity of heavy metals, arsenic, and boron has been demonstrated after accumulations in soils and subsequent uptake by various plants. Table 8-7 presents a summation of toxic effects of individual elements (Krupa et al., 1976). Published reports of direct effects on plants from specific sources are discussed in the following paragraphs.

Arsenic--Only one study was available to show foliar uptake of airborne arsenic (Linzon, 1977). Phytotoxicological studies in the vicinity of gold smelters in Ontario, Canada revealed the occurrence of several injuries to vegetation (primarily fireweed, Epilobium angustifolium) as induced by airborne arsenic compounds in a sulfurous plume. Chemical tissue analysis of affected leaves revealed arsenic at 200 ppm as compared to 1 ppm arsenic in leaves collected 50 miles from the source area. Linzon (1977) suggested 10 points of evidence leading to the conclusion of the airborne nature of the toxicant including elevated concentrations on sides of plants closer to the source and correlation of arsenic emissions with corresponding changes in tissue concentrations on an annual basis.

Arsenic sprays have been applied to the foliage of many plants to hasten fruit maturation by causing premature defoliation and chemical changes in the fruit. For example, lead arsenate sprayed on grapefruit trees caused a "fruit gumming" reminiscent of boron deficiency (Liebig, 1966). Boertitz et al. (1976) reported that arsenic deposited at 22 mg/kg on soil reduced the yield of wheat, rye, winter rape, and red clover by 25, 8, 0, and 6 percent, respectively.

Cadmium--Most biologically active cadmium enters plants through root uptake (Jordan, 1975). Small oxide particles (0.01 to 0.03 μm) may enter leaves through stomata, but it is thought that the oxides remain largely inert. Cadmium accumulated by apple leaves may be translocated and incorporated into fruits as they develop (Yopp et al., 1974).

Copper--Wu and Bradshaw (1972) demonstrated a selection of individual plants of the grass Agrostis stolonifera growing near metal smelters, thus indicating an indirect effect of within-species simplification within a population through selection.

Lead--Davis and Barnes (1973) reported reduced growth of loblolly pine and red maple seedlings in pots of two forest soils treated with 2×10^{-4} to 2×10^{-3} molar lead chloride. Lead toxicity symptoms may include fewer and smaller leaves, reduced plant size, leaf yellowing, and necrosis of alder, sugar beet, squash, and bush bean (Schoenbeck, 1973). Plants growing in soil already high in these metals tended to be more sensitive to the addition of metals by air pollution.

Nickel--Plants can absorb and translocate airborne nickel salts (NAS, 1975). Once inside the plant, nickel affects photosynthesis and other processes such as stomatal function (Bazzaz and Govindjee, 1974). In cases of incipient nickel toxicity to vegetation, no definite symptoms have been observed other than growth repression. In cases of moderate or acute nickel toxicity, chlorosis-resembling symptoms of iron deficiency are common (Anderson et al., 1973).

TABLE 8-[illegible]. PLANTS SENSITIVE TO HEAVY METALS, ARSENIC, AND BORON AS ACCUMULATED IN SOILS AND TYPICAL SYMPTOMS EXPRESSED

Metal	Plant	Symptoms	Reference
Arsenic	Snap bean, lima bean, [illegible], pea, cucumber, alfalfa, legumes, sweet corn, strawberry (on light and sandy soils).	Reduced germination of seeds, rotting of roots, leaf wilt, brown to red coloration of leaves, reduced yield in fruit trees, death.	Liebig (1966) Linzon (1977)
Boron	[illegible], var. [illegible]; [illegible], var. Henderson; kidney bean, var. [illegible]; [illegible], var. [illegible]; onion, var. [illegible]; pea, var. Alaska; peach, var. [illegible]; [illegible], var. Kaki; rose, var. Snow White; [illegible], var. [illegible]; wheat, var. Opal; yellow zinnia.	Yellowing of leaf tips, necrosis between lateral veins and midrib of monocotyledons, marginal leaf scorch, downward cupping of leaves, reduced flowering, fruit lesions.	Bradford (1966) Krupa and Kohut (1976) Yopp et al. (1974)
Cadmium	Red oak, birch, [illegible], beet, carrot, celery, green pepper, lettuce, radish, [illegible], tomato, winter wheat.	Reduced root elongation, general growth retardation.	Yopp et al. (1974) Jordan (1975)
Copper	Bean, citrus fruits, [illegible].	Stunted root development, chlorotic leaves, reduced vegetative growth.	Reuther and Labanauskas (1966)
Lead	Bean, [illegible], var. [illegible], beet, corn, fescue, lettuce, [illegible].	Stunted root growth, shoot retardation, increased leaf abscission, reduced yields.	Yopp et al. (1974)
Magnesium	Orange (only case known from published literature).	Leaf rolling, spiraling inhibition of leaf emergence. Narrow leaf development.	Embleton (1966)
Manganese	Alfalfa, [illegible], cabbage, cauliflower, cereals, citrus, clover, [illegible], tobacco, [illegible], barley, var. Atlas 46, var. [illegible], cranberry, peanut, potato, var. Kennebec; alfalfa, [illegible], apricot, barley, bean, brussels sprout, carrot, clover, [illegible], orange, pea, potato, sugar beet, vetch, [illegible].	Necrotic spots on leaves, necrosis of internal bark,marginal leaf yellowing, incurling of leaf margins.	Yopp et al. (1974) Labanauskas (1966) NAS (1973)
Mercury	[illegible], butterfly weed, cinquefoil, fern, [illegible], privet, sunflower, willow.	Possible reduced growth.	Yopp et al. (1974) Lagerwerff,(1972) Jacobson and Hill (1970)
Nickel	Citrus fruits, [illegible], oats, var. [illegible]; pear.	Depression of vegetative growth, leaf chlorosis, white or light yellow and green striping.	Vanselow (1966) Yopp et al. (1974)

TABLE 8-7. (continued)

Metal	Host	Symptom	Reference
Potassium	Orange, lemon ([illegible] for published literature).	Fruit coarseness and leaf necrosis, leaves curl downward, marginal leaf necrosis, interveinal chlorosis, plant dieback.	Ulrich and Ohki (1966)
Zinc	Oats, [illegible] barley var. [illegible]; corn, var. [illegible]; [illegible], var. [illegible], barley, citrus, oats, [illegible].	Uniform chlorosis, reduced terminal growth, twig dieback, chlorotic striping of leaves, stems stiff and erect.	Chapman (1966) Yopp et al. (1974)

Source: Adapted from [illegible] et al. ([illegible])

8.7.2 Classification of Plant Sensitivity--Particles

Coarse particles have not been shown to elicit responses in plants in such a manner as to allow plants to be placed into sensitivity classes similar to those developed for gaseous pollutants. Accumulations of particulate matter, such as roadside dusts, cement, quarry particle emissions, or other forms of deposits, such as fly ash, are deposited on all surfaces and induce responses discussed under the symptom portion of this chapter. However, heavy metals do elicit different responses in plants, and therefore it is possible to develop lists of particularly sensitive plants.

Heavy metals are constituents of many coarse particles emitted from various sources. To our knowledge, there has not been an organized effort to establish the toxicity of specific chemical constituents of particles in relation to sensitivity groups of vegetation under field conditions. Table 8-7 lists plants that may be sensitive to heavy metals following deposition and various symptoms as expressed following their respective accumulations.

8.8 EXPOSURE-RESPONSE RELATIONSHIPS--PARTICLES

Review of the published literature suggests that it is not possible at present to define, even crudely, exposure-response relationships for the effects of particulate air pollutants on plants. Many reports deal only with gross visible effects or tissue accumulation of one or more constituents of the atmospheric particulate matter present. The emphasis of research has been on settleable coarse particles. Since these are conglomerates of several pollutants, their chemical compositions are frequently ill-defined, although their sources have often been identified. Little information could be found on the effects of fine particles on vegetation.

Where cause-and-effect relationships have been suggested, generally no information is presented on the actual concentration, particle size, and frequency distributions. Deposition rates and plant effects vary significantly with particle size. Few studies are available where two independent scientists have evaluated the effects of particles on vegetation with closely comparable physical and chemical properties under reproducible conditions.

Much of the literature refers to particles from point and line sources and their accumulation in or on soils and vegetation. Tissue accumulation of a given element must be considered as a plant response. Soil scientists have contributed most of the information on plant toxicity symptoms that has been obtained under laboratory conditions. Since many of the plant effects observed are due to the accumulation of elements up to toxic concentrations, tissue concentrations prior to actual exposure will affect the amount and elemental composition of particles that plants can tolerate. Exposure-response relationships are determined by the concentrations of various elements in the soil where the plant is growing.

Effects of surface accumulation of cement kiln dust on bean leaves have been investigated by Darley (1966). Doses of 0.6-3.8 g/m per day were applied for 2 or 3 days, and foliar injury and reductions in carbon dioxide exchange were observed. Reductions in carbon dioxide exchange of up to 33 percent were noted in the absence of visible foliar injury. Bean leaves dusted with cement kiln dust at the rate of 4.7 g/m^2 per day for 2 days and then exposed to dew developed leaf rolling and intervenal necrosis (Lerman and Darley, 1975). Leaves not exposed to dew following the dust treatment remained asymptomatic.

Reduced yields and injury to leaves and flowers of several plant species were observed when the plants were exposed once a week for 4 weeks to a dust containing cadmium, lead, copper, and manganese (Krause and Kaiser, 1977). Yield reductions of up to 36 percent were noted.

Plants accumulate different elements at differing rates. Tissue concentrations of some elements are known to be significantly higher in the vicinity of a source for those elements in comparison with background or baseline concentrations. This elevated tissue concentration may be due to direct foliar uptake or uptake from the pollutant accumulations in the soil. In many cases, elevated tissue concentrations of a given metal or metalloid are not paralleled by visible injury.

Demonstration of injury symptoms on vegetation under field conditions as a result of accumulation of metals or metalloids is rare. The demonstrated cases are for strip mine wastes. Predicted effects from atmospheric deposition include plant community changes, chronic long-term physiological changes, and indirect effects through a modification of the response to other types of stress. Thus, the state of our knowledge concerning the effects of particles on vegetation is inadequate at this time and does not allow the development of accurate dose-response curves.

8.9 INTERACTIVE EFFECTS ON PLANTS WITH THE ENVIRONMENT--PARTICULATE MATTER

Few studies have examined the influence of dusts or heavy-metal-containing particles on the interactions between organisms capable of causing disease and the predisposition of the host plant to the disease process.

Infection due to *Cercospora* spp. increased on sugar beet leaves exposed to cement dust containing 36 percent calcium oxide and 15 percent silica (Schoenbeck, 1960). Increased occurrence of fungus-induced leaf spots on wild grape and sassafras has been observed near a source of heavy emissions of limestone dust (Manning, 1971). After examining 40 leaves in each of five locations exposed and not exposed to the dust accumulations, he found that disease development was two to three and six to seven times greater, respectively, for the two leaf spot fungi in the exposed areas.

Natural exposure to combustion nuclei from automobile exhaust, which supplied increased levels of Aitken nuclei and atmospheric lead, reduced germination of uredospores of *Puccinia* striiformis (stripe rust of wheat); in situ development of disease was prolonged about 4 days. Similar studies at an unexposed site did not reveal reduced spore germination (Sharp, 1967, 1972).

8.10 EFFECTS OF SULFUR DIOXIDE AND PARTICULATE MATTER ON NATURAL ECOSYSTEMS

The previous sections of this chapter have discussed the effects of sulfur dioxide and particulate matter on individual plants. This section discusses the effects of these substances on natural ecosystems since, because of their complexity, these systems respond to environmental perturbations differently from individuals or monocultures of organisms.

Ecosystems are basically energy-processing systems, the components of which have evolved together over a long period of time. They are composed of living organisms together with their physical environment (see Chapter 7, Table 7-1). The boundaries of the system are determined by the environmental conditions that determine the kinds of life forms that can exist in a particular habitat or region. The plant and animal populations within the system are the objects through which the system functions. Ecosystems respond to environmental changes or perturbations only through the response of the organisms of which they are composed (Smith, 1980).

Relationships among the various ecosystem components are structured, not haphazard. The living (biotic) and the nonliving (abiotic) units are linked together by functional interdependence. Processes necessary for the existence of all life, the flow of energy and the cycling of nutrients, are based on the functional relationships that exist among the organisms within the system (Odum, 1971; Smith, 1980; Billings, 1978). Because of these relationships, unique attributes emerge when ecosystems are studied that are not observable when individuals, populations or communities are studied. For a more detailed account of ecosystems, see Chapter 7, Section 7.1.2.

The discussion that follows emphasizes the response of terrestrial ecosystems to sulfur dioxide and particulate matter. Natural ecosystems are seldom, if ever, exposed to a single air pollutant. Therefore, the responses observed under ambient conditions cannot conclusively be attributed to a single substance such as sulfur dioxide or particulate matter alone.

8.10.1 Sulfur Dioxide In Terrestrial Ecosystems

Sulfur is an element that is essential for the normal growth and development of plants and animals. It is a basic constituent of protein and is required in large amounts by some plants. Under normal circumstances sulfur in rainwater and in soil organic matter is sufficient to meet plant requirements. Excessive sulfur in the form of sulfur dioxide, however, can be toxic to plants. The phytotoxic forms of sulfur, routes of entry into plants, and the symptomatology of SO_2 injury to plants have been discussed in the preceding sections.

Within any ecosystem, nutrient sources are to be found in the atmosphere, in living and dead organisms, and in available and unavailable salts in the soil and rocks. The nutrients are cycled from the living to the nonliving components and back again. Air pollution, however, can disrupt nutrient cycling by altering the amounts in the various compartments and the rate of flow among them (Smith, 1980).

There are two types of biogeochemical cycles, the sedimentary and the gaseous. The sulfur cycle includes both. Sulfur enters the atmosphere from the combustion of fossil fuels, from volcanic eruptions, from the surface of oceans and in gases released by decomposition processes (see Chapter 4). Anaerobic decomposition of organic matter releases hydrogen sulfide (H_2S) into the atmosphere where it is quickly oxidized into sulfur dioxide. Sulfur dioxide is soluble in water and may be carried back to earth in rainwater as dilute sulfuric acid (H_2SO_4) (Smith, 1980). Regardless of the source, sulfur in soluble form is taken up by vegetation and is incorporated through a series of metabolic processes, including photosynthesis, into sulfur-containing amino acids. Sulfur is transferred from the producers to the consumers in food and through excretion and death back to the soil and to the sediments in the bottoms of ponds, lakes, and seas where bacterial action releases it as hydrogen sulfide or as sulfate. Sulfur in the long-term sedimentary phase is tied up in organic and inorganic deposits in the soil, and sulfur is added to ecosystems through geological weathering and meteorological processes, with the latter being the predominant source. Weathering and decomposition permit sulfur to enter into solution and to be carried into aquatic and terrestrial ecosystems. In its gaseous state, sulfur is circulated on a global scale (Figure 8-3).

Based on empirical watershed studies (Likens et al., 1977; Shriner and Henderson, 1978) and modeling (Coughenour, 1978), the soil is a major reservoir for atmospherically derived sulfur within ecosystems. The majority of soil sulfur is unavailable to vegetation and is organically bound in the humus (May and Downes, 1968). Microbial activity oxidizes organically bound sulfur to sulfates. Sulfates may be taken up by plants or leached from the soil. The rate of sulfur released from the organic to the inorganic compartment is the major factor controlling the movement of sulfur between the soil and vegetation (May et al., 1972; Moss, 1976). A distinction between natural and agroecosystems is that soils in agroecosystems through continued cropping are depleted of their supply of organic sulfur, and it is not renewable; therefore, sulfur must be added as fertilizer. Sulfur dioxide brought down in precipitation is also added to the soil. The amount of sulfur added to soils through precipitation will depend on the industrial activity of the surrounding area (Kamprath, 1972).

The influence of anthropogenic sources of sulfur on the sulfur cycle is most pertinent when addressed on a regional basis (Granat et al., 1976) as Shinn and Lynn (1979) have done for the northeastern United States. Comparing global versus regional sulfur cycling, atmospheric sulfur additions are not equally distributed over the global land areas, and the northeastern United States experiences anthropogenic atmospheric additions of sulfur that are 28.4 times that expected if additions were distributed uniformly over the globe. The most notable contrast between the global and regional sulfur cycle is the importance of atmospheric sulfur sources. Globally, natural processes far exceed anthropogenic contributions, whereas in the northeastern United States human activities generate 12.5 times the amount of sulfur released by nature. A total of 27×10^6 tons of SO_4 enters the northeastern regional atmosphere annually and 13×10^6 tons are deposited within this region by wet and dry deposition; the remaining sulfur is exported to other areas of the globe.

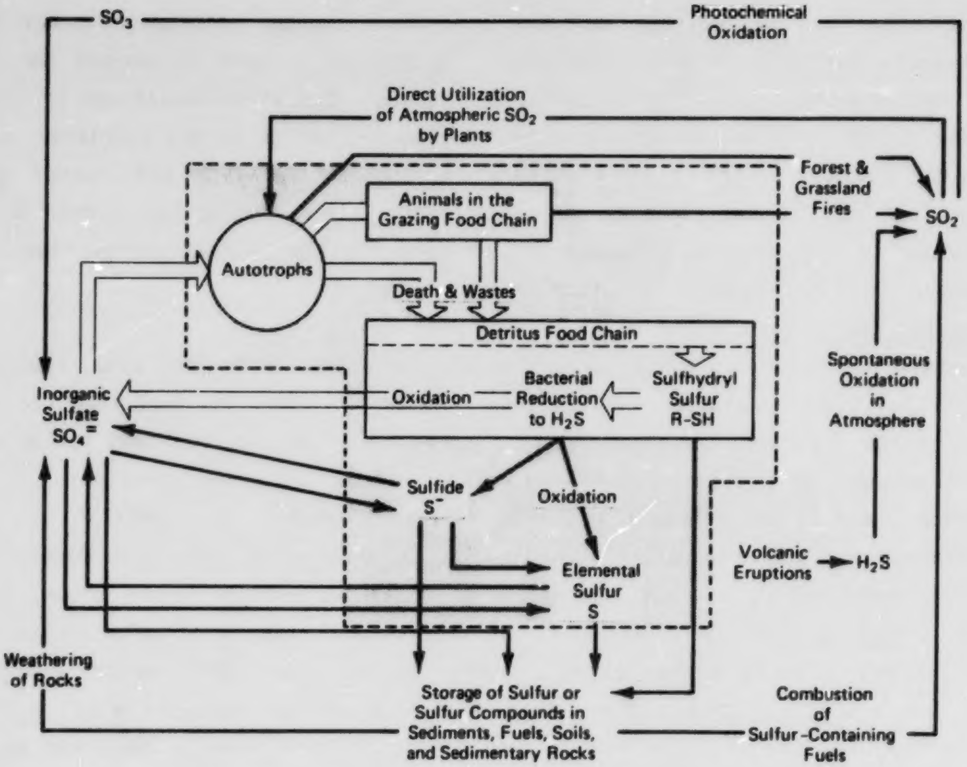
SO$_3$
Photochemical Oxidation
Direct Utilization of Atmospheric SO$_2$ by Plants
Forest & Grassland Fires
SO$_2$
Animals in the Grazing Food Chain
Autotrophs
Death & Wastes
Detritus Food Chain
Inorganic Sulfate SO$_4^=$
Oxidation
Bacterial Reduction to H$_2$S
Sulfhydryl Sulfur R-SH
Spontaneous Oxidation in Atmosphere
Sulfide S$^=$
Oxidation
Elemental Sulfur S
Volcanic Eruptions
H$_2$S
Weathering of Rocks
Storage of Sulfur or Sulfur Compounds in Sediments, Fuels, Soils, and Sedimentary Rocks
Combustion of Sulfur-Containing Fuels

Figure 8-3. The sulfur cycle (organic phase bounded by dashed line).

Source: Clapham (1973).

Based on the above information, man is clearly a major source of atmospheric sulfur within the United States and, in the Northeast alone, anthropogenic sources exceed all others by a factor of 12.5. Within this region, SO_2 levels annually average 16 $\mu g/m^3$ (0.006 ppm) (Shinn and Lynn, 1979), which is several times that recorded in pristine areas. The immediate fate of approximately 60 percent of this atmospheric sulfur is deposition on terrestrial and aquatic ecosystems; however, the subsequent fate of sulfur within the ecosystem is not fully known. Experimental evidence from forested watersheds coupled with results from simulation models indicates that sulfur in ecosystems is highly mobile. Although sulfur levels in the soil and vegetation compartments in aggrading and mature forest ecosystems impacted by SO_2 increase with time, the majority of sulfur deposited annually is exported out of the system in stream flow.

The conclusion that SO_2 emitted into the atmosphere through anthropogenic activity is ultimately transferred to terrestrial and aquatic ecosystems is well documented (Meszaro et al., 1978). Unfortunately, the fate of sulfur in the ecosystem after deposition is not fully resolved. The issue is critical because ecosystems subjected to excess nutrients or toxic materials do not commonly distribute them uniformly throughout the system but, rather, preferentially sequester them in specific pools or compartments. In addition, sulfur dioxide as a gas can cause injury to the vegetative components of an ecosystem so that energy flow and the cycling of other nutrients as well as sulfur may be disrupted.

8.10.2 Ecosystem Response to Sulfur Dioxide

Pollutants often reduce community diversity when an ecosystem becomes simplified through the removal of pollutant-sensitive plant species. Woodwell (1970) has described the effects of pollution on the structure and physiology of ecosystems. The studies that follow discuss the ecological changes in a forested ecosystem resulting from sulfur gas emissions.

The Kaybob I and II gas plants (Fox Creek, Alberta, Canada), which emit SO_2 during the removal of hydrogen sulfide from natural gas, are located within the transition Montane-Boreal forest that is dominated by a mixed assemblage of deciduous and coniferous trees; however, white spruce (Picea glauca) is predominant in successional stands on well-drained soils. (Winner and Bewley, 1978a,b; Winner et al., 1978). Because these white spruce forests have less species variation than other sites, they were selected for analysis along a transect showing decreasing SO_2 stress. The Kaybob facility began operation in 1968, and the field study was conducted during the summers of 1975 and 1976. From 1973 to 1975, it is estimated that the Kaybob facility emitted approximately 71,000 kg per day of SO_2 (Winner and Bewley, 1978b).

Relative species diversity showed no gradient pattern of response to SO_2; however, the percent of coverage by all understory plants, including vascular plant species and mosses, showed a marked increase with distance from the source (Winner and Bewley, 1978b). White spruce seedlings close to the refinery were reduced in number. Changes in moss communities were conspicuous and included decreasing values for moss canopy coverage, moss carpet depth, dry weight, and capsule number and for the frequency of physiologically active versus inactive moss plants. Close to the source, there were no mosses at all (Winner and Bewley, 1978a). These results indicate that species diversity, particularly in the mosses, has changed as a consequence of sulfur gas emissions. The changes in diversity observed during the summers of 1975 to 1976 probably were initiated soon after the facility began operation in 1968 and the effects will be observed as long as the facility is operational. Atmospheric SO_2 concentrations were not reported, but it is evident that operational upsets, particularly during the early years of plant operation, resulted in high emission rates from the low elevation flare sources causing high SO_2 concentrations near the ground in this area.

In a subsequent study, Winner et al. (1978) used sulfur isotope ratios to determine the fate of sulfur emissions from the natural gas plants. The $\delta^{34}S$ value of SO_2 emitted from gas refineries is higher than that for nonindustrial sources in the study area. Through the use of this tracer, it was determined that mosses absorb only airborne sulfur, whereas the sulfur in conifer needles is derived from both air and soil. Krouse (1977) in his study of the Ram River area of Alberta made a similar determination. Winner et al. (1978) conclude that because mosses act as a sink and accumulate large quantities of sulfur, they are more vulnerable than vascular plants to effluents such as SO_2. Further, they state that mosses participate in the process of soil formation, prevent erosion, and play a role in succession. Removal of the moss carpet would ultimately be expected to influence the growth of the vascular plants in the ecosystem because when mosses are not present, soils accumulate SO_2 rapidly.

In another, more detailed study of the effects of low-level sulfur dioxide emissions on a forested ecosystem, it was observed that the main ecological process that was directly and indirectly affected by the sulfur gas emissions was nutrient cycling (Legge et al., 1981, Legge 1980). The study concluded that the chronic exposure to sulfur gas emissions over time resulted in an alteration of the mineral nutrient balances of the various ecosystem components and modified the biological relationships among the components. Further, the four-year study, begun in 1972 concluded that despite measurable deterioration of the forest ecosystem in the West Whitecourt study area, it did not appear that the sulfur gas emissions caused irreversible ecological degradation, particularly in view of the significant reductions in sulfur gas emissions that have resulted since the inception of the West Whitecourt Gas Plant situated in west central Alberta, Canada (Legge et al., 1981; Legge, 1980).

The region studied by Legge et al. (1981) was being subjected to the effluents from the West Whitecourt "sour gas" processing plant. "Sour gas" is natural gas (methane) that contains hydrogen sulfide (H_2 S). For commercial use, the hydrogen sulfide is removed from the methane as elemental sulfur by a chemical process. Any H_2S not converted to elemental sulfur is incinerated in excess air and methane in a high temperature (580°C) furnace and oxidized to SO_2, which is then vented to the atmosphere from a tall (122 m) incinerator stack. Smaller flare stacks are used to vent the effluents from the burning of small waste quantities of sulfur recovery gas plant process and compressor gases. The incinerator stack is the main source of SO_2 emissions, except when operational upsets occur within the gas plant. At these times, the flare stacks may for short periods of time contribute more gas emissions to the atmosphere daily than the incinerator stack. Since the gas plant began functioning in 1961-62, the SO_2 output has dropped from 600 to 36 metric tons per day in 1976 because of improved operating procedure.

The vegetation of the Whitecourt study area is within the predominantly forested subregion of the Boreal Forest of Canada and is characterized as a transition forest area between the Boreal and Subalpine Forest Regions. The tree populations in the area are hybrid representatives of lodgepole pine x jack pine (Pinus contorta Loud. x P. banksiana Lamb.) and of the alpine fir x balsam fir [Abies lasiocarpa (Hook.) Hutt. x A. balsamea (L.) Mill.]

To determine the effect of low-level sulfur dioxide emissions on the forested ecosystem, an integrated ecological approach was used to study and explain the changes in ecosystem structure and function. The four-year study, begun in 1972, divided the ecosystem into four major components (air, vegetation, soil, and water) and developed a conceptual model to illustrate the dynamic relationship between the sulfur dioxide "source" and the generalized ecosystem "sink" (Legge et al., 1981; Legge, 1980).

Five experimental sites were located, sequentially, downwind in the main path of the sulfur dioxide emissions. The two intensive experimental sites were located 1.0 and 1.5 km east of West Whitecourt Gas Plant. Analysis of the ambient air monitoring data from the two intensive sites during the 1975 and 1976 growing seasons revealed that the air quality standard for sulfur dioxide of 0.2 ppm per half hour per 24 hours as set by the Alberta Department of the Environment was exceeded on only three occasions at the two sites in over 2500 hrs of monitoring. Much higher concentrations existed at this sampling site from 1961-74, however, and effects on soil pH, nutrient balance and tree growth would have also been influenced by these higher concentrations. In addition, it was observed that the stable sulfur isotopic composition ($^{32}S/^{34}S$) of the sulfur dioxide emissions from West Whitecourt Gas Plant was different from the natural background sulfur and so provided a tracer that indicated that the gas plant was the major source of the sulfur gas detected in the forest ecosystem (Legge et al., 1981, Legge 1980).

The gas exchange properties of field acclimated lodgepole x jack pine trees were measured to determine the short- and long-term effects of chronic exposure to sulfur dioxide. Photosynthetic rates and leaf resistance of the trees had been modified. The extent of ecolo-gical modification was dependent on the distance from the sulfur source. The rates of photosynthesis were lower and leaf resistance rates were higher nearer the source of sulfur dioxide. The reduced rate of photosynthesis was only partially attributable to increased leaf resistance; therefore, ecological factors such as the mineral nutrient status of foliage and soil pH were studied since these parameters were known to modify plant response (Legge et al., 1981).

Analysis of the foliage of the lodgepole x jack pine trees indicated that the sulfate-sulfur concentration decreased with increasing distance from the sulfur dioxide emission source. The decrease in foliar sulfate concentration was also more pronounced with increased foliar age. In addition, detailed analysis of foliar mineral nutrient concentrations revealed that the mineral nutrient status of the trees had been altered. The concentration of P, K, Fe, Mg, N, and Zn in the foliage tended to increase with distance from the West Whitecourt Gas Plant while the concentration of Ca and Al tended to decrease. Much more than any of the other minerals sampled, the Mn concentration in the foliage tended to decrease as the distance from the sulfur source increased. Reduction in foliar potassium and phosphorus may be associated with reduction in the rate of photosynthesis since foliar potassium has been linked with stomatal activity and phosphorus with phosphorylation. The alteration of mineral nutrients in the foliage of the lodgepole x jack pine trees is, therefore, an important ecological factor contributing to the modification of plant response by sulfur dioxide emissions (Legge et al., 1981 and Legge 1980).

Soil pH is known to affect the availability of mineral nutrients to plants; therefore, soil pH profiles were measured. Soil pH and soil carbonate values increased with depth in 100 cm profiles and with distance away from the West Whitecourt Gas Plant. Total sulfur in the soil likewise decreased with depth and distance from the emission source, suggesting that sulfur is tied up in the organic matter and released very slowly. No relationship was found between soil pH and $\delta\ {}^{34}S$ values in the soil; however, the soil $\delta\ {}^{34}S$ value was found to be an excellent indicator of the presence and penetration into the soil profile of sulfur originating from sulfur dioxide emissions, while soil pH indicated sulfur loading.

A direct relationship was found between lowered soil pH and the elevated levels of foliar Mn in lodgepole x jack pine trees. This relationship suggests that the foliar Mn concentration could be used as an indicator of mineral nutrient modification of the forest ecosystem by sulfur dioxide emissions.

Biochemical changes in the trees were also noted. The most significant was a transient metabolic effect involving adenosine triphosphate (ATP). ATP was found to decrease significantly in the foliage of the trees when they were exposed to low concentrations of SO_2 for a short duration. Foliar ATP concentrations returned to the pre-SO_2 fumigation levels when the fumigation ceased. Trees grown in the laboratory in the absence of sulfur dioxide emissions showed no fluctuation in foliar ATP concentrations and the ATP concentration was three times the content of field-grown trees. The lower concentration of ATP in the foliage of field-grown trees when compared to laboratory-grown trees suggests a partial explanation for the lowered photosynthetic capacities of the lodgepole x jack pine trees growing in the Whitecourt area. The short-term reversible change in foliar ATP concentration observed in the foliage of the hybrid pine trees chronically exposed to sulfur gas emissions in the field is suggested as a biochemical response to SO_2 toxicity.

It was assumed that the observed ecological modifications of the forest ecosystem, such as reduced needle biomass (premature needle drop), reduced biochemical energy, reduced photosynthetic rates, reduced soil pH, disruption of mineral nutrient cycling, and foliar sulfur loading, when combined with a shortened growing season, should be measurable as a reduction in forest production. There was a definite reduction in annual basal area increments in the lodgepole x jack pine trees since 1962 that was attributed to sulfur dioxide emissions from the West Whitecourt Gas Plant indicating that ecological modifications were initiated long before the study began in 1972. The maximum reduction in basal area increments occurred nearest the plant and the reduction fell to zero at 9.6 km. The area affected by sulfur dioxide during the past 14 years is estimated at 454 km^2 (175 mi^2) or 45,373 ha (112,130 acres). The significant reduction in sulfur dioxide emissions since 1970 suggests areas being impacted by the sulfur dioxide will decrease in the future, thus permitting the recovery of the portion of the ecosystem no longer exposed to sulfur dioxide stress.

In the near future, the grasslands of the upper plains will be subject to SO_2 emissions from new coal-burning power facilities that are being constructed in areas rich in coal reserves (Durran et al., 1979; Preston and Lewis, 1978). To address this problem, plots of Montana grasslands were exposed to SO_2 during growing seasons of successive years. The monthly median exposure levels were approximately 0, 52 μg/m^3 (0.02), 106 μg/m^3 (0.04), and 185 μg/m^3 (0.07 ppm) SO_2 and were delivered by a zonal air pollution system or ZAPS (Lee et al., 1978). Field observations over four years verified that these concentrations were not sufficient to elicit any leaf lesions characteristic of acute SO_2 injury (Heitschmidt et al., 1978). Ambient pollutant concentrations were typically greater at night, and the concentration decreased rapidly from the interface of turbulent air and grass canopy downward to the soil (Preston, 1979).

The most prevalent producer species within the grassland is a perennial, Agropyron smithii. In populations sampled over the growing season in each of the exposure regimes, SO_2 induced a variety of changes in biochemical indices of plant performance. Monthly samples of tillers and leaves showed a positive correlation of foliar sulfur with time of exposure and canopy-level SO_2 concentrations (Lauenroth and Heasley 1980). This relationship was most conspicuous with the two higher exposure regimens, and total foliar sulfur in the highest exposure plot was three times greater than that in vegetation sampled from control locations (Lauenroth and Heasley, 1980). As the sulfur content of leaf tissue increased, the ratio of nitrogen to sulfur decreased (Lauenroth and Heasley, 1980).

These biochemical changes in the major producer species were mirrored by other modifications in plant performance. In A. smithii populations exposed to 52 μg/m^3 (0.02 ppm) SO_2 over the growing season, the functional leaf life (the period of active photosynthesis) was increased by several weeks, while the same index of plant performance was shortened by two weeks at 106 (0.04 ppm) and 185 μg/m^3 (0.07 ppm) SO_2 (Lauenroth and Heasley, 1980). Parallel increases and decreases in chlorophyll content at the low and high SO_2 levels, respectively, were also recorded; however, the measurements of net primary production from harvested samples over a period of 5 years revealed no significant treatment response. Finally, the single significant crop change noted was a decrease in the carbon stored in the rhizomes of western wheatgrass (Lauenroth and Heasley, 1980).

Dominant producers were not the only flora exhibiting sensitivity to SO_2. In simulated pollutant exposure using a bisulfite solution, Sheridan (1979) showed that nitrogenase activity in a major component of the lichen flora (Dollema tenex) was reduced. Although the applicability of the data must be validated through field studies, the potential for such an effect must be recognized, particularly in the light of the importance of soil lichens in regulating nitrogen fixation in the grasslands (Sheridan, 1979).

Further evidence of SO_2-associated effects on grasslands is recorded in both consumer and decomposer populations. The density of grasshoppers, a major consumer of A. smithii foliage, decreased 25 percent in two successive seasons with increasing SO_2 stress (Lauenroth and Heasley, 1980). Decomposition rates were apparently also altered, with less litter disappearance in SO_2-exposed plots. The mechanism involves a direct pollutant effect on decomposer activity rather than an indirect effect, such as an increased sulfur concentration in the litter (Lauenroth and Heasley, 1980).

Larger consumers also exhibited responses reflecting the presence of SO_2 in the atmosphere; however, the responses were not dose-dependent (Chilgren, 1978). Peromyscus maniculatus, prairie deer mouse, is a common and active vertebrate in grassland communities. Over one exposure season, the frequency of P. maniculatus in control plots increased, implying an SO_2-induced behavioral response (habitat preference) whereby individuals seek habitats free of the pollutant.

In summary, at levels above 52 $\mu g/m^3$ (0.02 ppm), SO_2 induced changes in the performance of producers, consumer, and decomposers. Many of the responses are individually small, but collectively, over time, they are gradually modifying the structure and function of the grasslands. The significance of these changes to the long-term persistence of the ecosystem remains controversial (Preston, 1979). This is particularly true since plots were not replicated in the Montana studies; on the other hand, the fact that the results are based on data accumulated over several years tends to add to its credibility.

The results of these studies, particularly the West Whitecourt and Montana grasslands studies, document the usefulness of addressing ecosystem-level responses to SO_2 from a multidisciplinary approach incorporating investigations of physiology, autecology, synecology, geochemistry, meteorology, and modeling. The results confirm that producers are sensitive to direct SO_2 effects as evidenced by SO_2-associated changes in cell biochemistry, physiology, growth, development, survival, fecundity, and community composition. Such responses are not unexpected. An equally important point of agreement among the different research efforts is the potential for ecological modification resulting from either direct SO_2 effects on nonproducer species or direct changes in habitat parameters, which in turn affect an organism's performance. Changes in biogeochemistry, particularly in the soil compartment, are notably responsive to chronic SO_2 exposure.

The influence of prolonged SO_2 exposures on plant communities is not well documented; however, a theoretical basis is emerging from which to evaluate effects. This conceptual effort is exemplified by the development of a generalized forest growth model (Botkin et al., 1972) that was designed to assess the consequences of the long-term interactions of air pollution stress and forest community dynamics (Botkin, 1976; Shugart and West, 1977). This model has been applied to determine the response of a mixed-species deciduous forest in the southeastern United States to the differential levels of growth reduction (0, 10, and 20 percent) occurring following simulated air pollution stress (West et al., 1980). Over several decades, simulated pollution stress altered the biomass importance of the major tree species within the forest; some species populations increased, while others decreased in importance. These results suggest that competitive interactions among species may significantly modify both the level and direction of change in growth rate of individual species in response to air pollution stress. Stand age was also shown to influence strongly the role of competition in modifying responses of individual species within the forest community. Since community composition is determined in part by species interactions (e.g., competition, symbiosis), the ecological importance of resistant species, the prominence of which in the community is determined by interaction with sensitive species, can be expected to be enhanced under stresses such as air pollution that do not affect all species equally (West et al., 1980). An understanding of the governing role of species interactions is essential for predicting how ecosystems will respond when exposed to low concentrations of pollution (Botkin, 1976). This is also the justification for not freely extrapolating the results from intensely managed forest and agroecosystems to predict how a mixed species community (e.g., natural forests or grasslands) will respond to a comparable perturbation (Miller and McBride, 1975; Kickert and Miller, 1979).

The results from community-level studies in areas experiencing chronic levels of SO_2 lend credibility to the modeling effort. Using communities composed of only 2 or 3 different species, Guderian (1967) analyzed community-level responses to SO_2 and their underlying causes. Changes in community composition were a function of pollutant dose; the higher the dose, the more rapidly the community changed. Altered community composition was attributed both to direct SO_2 effects on populations of sensitive species and to indirect changes in species interactions. Community biomass exhibited little quantitative change, but striking differences in species composition. Similar conclusions have been reached in studies of natural plant communities experiencing prolonged SO_2 exposure (Guderian and Stratmann, 1968; Rosenberg et al., 1979). Rosenberg et al. (1979) assessed the species composition in 27 stands of a natural regrowth of a Northern hardwood forest dominated by oaks (Quercus spp.), white pine (*P. strobus*), and hemlock (*Tsuga canadensis*). The stands, which varied in their distance from a 25-year-old coal-consuming power plant, exhibited no obvious *a priori* compositional differences. Atmospheric pollutant levels were not reported, although foliar symptoms typical of SO_2 toxicity were recorded on several occasions. In both upwind and downwind directions, the number of vascular plants (canopy, understory, and ground) per unit area (species richness) increased with distance from the source; however the differences were greater downwind. A similar distance-dependent response was also recorded for species diversity using the Shannor-Weiner index. For both species richness and species diversity, the rates of increase were more gradual downwind from the power plant. In spite of these SO_2-induced changes in community composition, an index of above-ground biomass (basal area of overstory species) exhibited no variation among stands. Among the vascular plants, shrub and ground vegetation was more sensitive (diversity and richness) than the overstory to SO_2 stress. This susceptibility of the lower strata was attributed to the intense competition among individuals at an early phase in their life history when they are more sensitive, and to microhabitat factors that tend to increase SO_2 levels close to the ground.

Some of the most notable examples of SO_2 affecting plant communities are the responses of cryptogamic flora (lichens and mosses), and several reviews are available (DeSloover and LeBlanc, 1968; Hawksworth, 1971). A map of epiphytic lichen communities for England and Wales has been devised that associates progressive shifts in species composition with SO_2 levels (Hawksworth and Rose, 1970). In general, the higher ambient SO_2 levels were consistently associated with fewer species and an increasing relative frequency of crustose versus foliose or fruticose forms. The fidelity with which community composition changes in accordance with SO_2 has led to the suggestion that analyses of lichen communities be used as a bioassay to estimate ambient SO_2 levels.

Similar mapping efforts are reported for several regions of North America. In a rural area of Ohio surrounding a coal-consuming power station (emitting 930 metric tons SO_2 per day), the distribution of two corticose lichens, *Parmelia caperata* and *P. ruderta*, was markedly affected by high SO_2 concentrations (Showman, 1975). In regions experiencing an annual SO_2 average exceeding 50 $\mu g/m^3$ (0.020 ppm), both species were absent. The distribution of more resistant lichens was not noticeably affected until SO_2 levels exceeded 65 $\mu g/m^3$ (0.025 ppm) (annual average). Somewhat lower levels were projected by LeBlanc and Rao (1973, 1975) to affect the ability of sensitive lichen species to survive and reproduce. Acute and chronic symptoms of SO_2 toxicity in epiphytic lichens occurred when long range (May-October) average concentrations exceeded 80 $\mu g/m^3$ (0.03 ppm) and were between 16-80 $\mu g/m^3$ (0.006-0.03 ppm), respectively.

The susceptibility of cryptogamic flora to elevated levels of SO_2 may influence the movement of materials within the ecosystem. In the northwestern coniferous forests, lichens fix 2-11 kg/ha of nitrogen annually, which represents 5-20 percent of the total nitrogen requirement for the dominant producer, Douglas fir (Denison, 1973).

Because the functioning of all ecosystems is due to a network of biotic and abiotic interactions, it may be hypothesized that the effects of SO_2 on producers must have repercussions at other trophic levels. Demonstration of such responses, however, is difficult experimentally, and an accurate assessment of the specific importance of SO_2 in eliciting these responses is complicated by the often complex relationships among producers, consumers, and decomposers.

Consumers and decomposers may respond to SO_2 via a direct, adverse effect of the pollutant. The presence of elevated atmospheric levels of SO_2 is particularly relevant to soil organisms (Babich and Stotzky, 1974). This focus on soil-borne organisms takes on relevance since the rhizosphere is not only biologically active but also the major site for sulfur accumulation within the ecosystem (Legge et al., 1976). In a forested area experiencing atmospheric SO_2 levels averaging 126 $\mu g/m^3$ (0.048 ppm), the species composition of soil microflora shifted toward a greater number and frequency of species capable of utilizing the soil sulfur additions (Wainwright, 1979). Specifically, the levels of thiobacilli and sulfur-oxidizing fungi were positively correlated with levels of SO_2 stress and soil depth.

The edaphic and climatic environments strongly influence the community of plants, animals and microorganisms that develop at a given site. In natural ecosystems in sulfur-deficient soils, communities have evolved within the constraints imposed by a limited supply of sulfur. Although atmospherically derived sulfur may not be sufficient to cause injury, the prolonged input of sulfur may relax the constraints of a limited sulfur supply, thereby inducing shifts in species composition.

8.10.3 Response of Natural Ecosystems to Particulate Matter

Particulate matter originating from both natural and anthropogenic emission sources is a common component of the atmosphere. As discussed in Section 8.7, the heterogeneous physical and chemical nature of particulate matter presents problems in addressing the significance of elevated atmospheric particulate levels for natural ecosystems and agroecosystems.

Wet and dry deposition are the two processes by which particles are transferred from the atmosphere to terrestrial ecosystems. The fate of particles deposited on foliar surfaces depends on the solubility of the constituents (chemicals and elements), the occurrence of precipitation, and the sorptive capacity of the leaf (Little, 1973). Furthermore, many elements commonly associated with particles are essential for plant metabolism (e.g., zinc and phosphorus), and, as a consequence, absorption may be a means by which the plant can supplement its nutrient supply (Kloke, 1974). Leaf surfaces may be only a transitory site for particulate matter and its associated constituents. If not retained by the leaf, material is ultimately transferred to the forest f[illegible] through washoff in rain events. The net effect of these processes is to funnel le[illegible] deposits to the litter-soil complex. This conclusion is verified for many atmospherically derived heavy metals deposited in natural ecosystems (e.g., Coughtrey et al., 1979; Thompson et al., 1979).

Given the regional character of particulate emissions, particularly along the East Coast (U.S. EPA, 1978b), the fate of particulate matter in terrestrial ecosystems experiencing low levels of particulate pollution needs to be assessed. In a deciduous forest in the Southeast, wet and dry deposition of aerosols, gases, precipitation and large particles were major sources of trace element input to the forest floor, including 99 percent for lead, 44 percent for zinc, 42 percent for cadmium, 39 percent for sulfate and 14 percent for manganese (Lindberg et al., 1979). These seemingly large percentages are typical for rural or remote areas even though three major coal-consuming power facilities (total coal consumption of 8×10^6 metric tons per year) were within 20 km of the forest studied.

Irrespective of the source of particles deposited in the forest, the atmosphere contributed a major portion of the trace element inputs (Lindberg et al., 1979). Water solubility was critical, since insoluble constituents associated with the particulate matter were not readily mobilized within the forest. Any event promoting solubilization (e.g., aerosol formation, rainfall scavenging, moisture formation on leaves) enhanced an element's mobility.

The leaf surface is not the only accumulation site for particles and their associated constituents within the ecosystem. Through precipitation scavenging of particles in air, washoff of surface deposits, or litterfall, particles are transferred to the soil where they are tightly bound to decaying organic matter. The upper soil horizon, including the decaying organic material, is a region of intense biological activity as a result of the physical degradation of litter, remineralization of the bound materials and root uptake of the plant-available nutrients. Consequently, particulate emissions that interfere with microbial activity can have delayed effects on primary production (Tyler, 1972) and soil consumer species.

In summary, even though the impact of particulate matter on terrestrial ecosystems is most apparent near large emission sources, ecosystems within the same geographic region may be the site of deposition. Foliar surfaces are the most common site for initial dry and wet deposition; however, most material is eventually transferred to the soil. Particulate matter alone constitutes an ecological problem only where deposition rates are high. However, concern for terrestrial ecosystems must also address elements and chemicals that may be associated with the particulate matter. Solubility of these particulate constituents is a critical factor since insolubility limits mobility within the ecosystem. One common behavior of particles is their tendency to accumulate selectively within a given component of the landscape. Soils are long-term sites for the retention of many constituents found in particles. While this accumulation in the soil-litter layer has had demonstrable adverse consequences for ecological processes such as decomposition, mineralization, nutrient cycling and primary production around some point sources, the much lower levels chronically deposited over large regions have not yet produced documented adverse impacts on natural ecosystems.

8.11 SUMMARY

Sulfur dioxide and particulate sulfate are the main forms of sulfur in the atmosphere. Of the two, sulfur dioxide is potentially more injurious, particularly when it is in combination with other airborne pollutants. Plants may be exposed to these pollutants in several different ways. Dry and wet deposition of gases and particles may bring sulfur compounds into contact with plant surfaces and/or the soil substrate. The effects of such external exposures on plants are more difficult to assess than those associated with the entry of SO_2 through the stomata. Plant response to pollutant uptake may be influenced by such dynamic physical factors as light, leaf surface moisture, relative humidity and soil moisture. Such factors influence internal physiological conditions in plants as well as stomatal opening and closing and, therefore, play a major role in determining the sensitivities of the plant species or cultivars.

Sulfur dioxide must enter a plant through leaf openings called stomata to cause injury. Sulfur dioxide, after entering plant cells through the stomata, is converted to sulfite and bisulfite, which may then be oxidized to sulfate. Sulfate is about one-thirtieth as toxic as sulfite and bisulfite. As long as the absorption rate of SO_2 in plants does not exceed the rate of conversion to sulfate, the only effects of exposure may be changes in opening or closing of stomata or insignificant changes in the biochemical or physiological systems. Such effects may abate if SO_2 concentrations are reduced. Both negative and positive influences on crop productivity have been noted following exposures to low concentrations.

Symptoms of SO_2-induced injury in higher plants may be quite variable since response is governed by pollutant dose (concentration x duration of exposure), conditions of the exposure (e.g., day vs night, peak vs long-term), physiological status of the plant, phenological stage of plant growth, environmental influences on the pollutant-plant interaction, and the environmental influences on the metabolic status of the plant itself.

There are several possible plant responses to SO_2 and related sulfur compounds: (1) fertilizer effects appearing as increased growth and yields; (2) no detectable response; (3) injury manifested as growth and yield reductions, without visible symptom expressions on foliage or with very mild foliar symptoms that would be difficult to perceive as being induced by air pollution without the presence of a control set of plants grown in pollution-free conditions; (4) injury exhibited as chronic or acute symptoms on foliage with or without associated reduction in growth and yield; and (5) death of plants and plant communities.

A number of species of plants are sensitive to low concentrations of SO_2. Some of these plants may serve as bioindicators in the vicinity of major sources of SO_2. Even these sensitive species may be asymptomatic, however, depending on the environmental conditions before, during, and after SO_2 exposure. Various species of lichens appear to be among the most sensitive plants.

As the exposure to SO_2 increases, plants may develop more predictable and more obvious visible symptoms. Foliar symptoms progress from chlorosis, or other types of pigmentation changes, to necrotic areas. The extent of necrosis increases with exposure. Studies of the effects of SO_2 on growth and yield have demonstrated a reduction in the dry weight of foliage, shoots, roots, and seeds, as well as a reduction in the number of seeds. At still higher exposures there are further reductions in growth and yield. Extensive mortality has been noted in forests continuously exposed to SO_2 for many years.

The amount of sulfur accumulated from the atmosphere by leaf tissues is influenced by the amount of sulfur in the soil relative to the sulfur requirement of the plant. After low-level exposure to SO_2, plants grown in sulfur-deficient soils have exhibited increased productivity. Plant growth and development represents an integration of cellular and biochemical processes. The response of a given species or variety of plants to a specific air pollutant cannot be predicted on the basis of the known response of related plants to the same pollutant. Neither can the response be predicted by a given response of a plant to similar exposures to different pollutants.

Each plant is a different individual genetically, and therefore its genetic susceptibility and the influence of the environment at the time of exposure must be considered for each plant and each pollutant. The data presented in this chapter exemplify the fact that each plant is a separate entity, and, because of the variation in response shown by the different plant species and different cultivars of the same species, making generalizations is difficult. With this in mind, this chapter concludes that in arid regions, some species of vegetation would probably not show visible signs of SO_2 injury even at concentrations as high as 11 ppm for 2 hours. On the other hand, in many nonarid regions where environmental conditions such as high temperature, high humidity, and abundant sunlight enhance plant responsiveness to SO_2 exposure, many species of sensitive and intermediately responsive vegetation would likely, from time to time, show visible injury when exposed to peak (5 minutes), 1-hour, and 3-hour SO_2 concentrations in the range of 2600-5200 $\mu g/m^3$ (1-2 ppm), 1300-5200 $\mu g/m^3$ (0.5-2 ppm), and 790-2100 $\mu g/m^3$ (0.3-0.8 ppm) respectively.

In general, the studies cited in this chapter indicate that regardless of the type of exposure and the plant species or variety, there is a critical SO_2 concentration and a critical time period after which plant injury will occur. This plant response appears to be associated with the capability of a plant to transform within the leaf toxic SO_2 and $SO_3^=$ into the much less toxic $SO_4^=$ and ultimately to transfer or break down $SO_4^=$.

At present, data concerning the interactions of SO_2 with other pollutants indicate that, on a regional scale, SO_2 occurs at least intermittently at concentrations high enough to produce significant interactions with other pollutants, principally O_3. A major weakness in the approach to pollutant interactions, however, is the lack of in-depth analysis of existing regional air quality data sets for the three principal pollutants (SO_2, O_3, and NO_2). These data should determine how frequently and at what concentrations the pollutants occur together both spatially and temporally within regions of major concern. The relative significance of simultaneous versus sequential occurrence of these pollutants to effects on vegetation is also not well documented and is critical in evaluating the likelihood and extent of potential pollutant interactions under field conditions.

A few studies have reported that combinations of particulate matter and SO_2, or particulate matter and other pollutants, increase foliar uptake of SO_2, increase foliar injury of vegetation by heavy metals, and reduce growth and yield. Because of the complex nature of particulate pollutants, conventional methods for assessing pollutant injury to vegetation, such as dose-response relationships, are poorly developed. Studies have generally reported vegetational responses relative to a given source and the physical size or chemical composition of the particles. For the most part, studies have not focused on effects associated with specific ambient concentrations. Coarse particles such as dust directly deposited on the leaf surfaces can result in reduced gas exchange, increased leaf surface temperature, reduced photosynthesis, chlorosis, reduced growth, and leaf necrosis. Heavy metals deposited either on leaf surfaces or on the soil and subsequently taken up by the plant can result in the accumulation of toxic concentrations of the metals within the plant tissue.

Natural ecosystems are integral to the maintenance of the biosphere, and disturbances of stable ecosystems may have long-range effects which are difficult to predict. Within the United States, anthropogenic contributions to atmospheric sulfur exceed natural sources. In the Northeast these contributions exceed natural sources by a factor of 12.5, and approximately 60 percent of the anthropogenic emissions into the atmosphere are deposited (wet and dry deposition) on terrestrial and aquatic ecosystems. The subsequent fate and distribution of sulfur in these systems is not well understood. Wet deposition of sulfur compounds is discussed in Chapter 7.

Data relating ecosystem responses to specific doses of SO_2 and other pollutants are difficult to obtain and interpret because of the generally longer periods of time over which these responses occur and because of the many biotic and abiotic factors that modify them.

Vegetation within terrestrial ecosystems is sensitive to SO_2 toxicity, as evidenced by changes in physiology, growth, development, survival, reproductive potential and community composition. Indirect effects may occur as a result of habitat modification through influences on litter decomposition and nutrient cycling or through altered community structure. At the community level chronic exposure to SO_2, particularly in combination with other pollutants such as O_3, may cause shifts in community composition as evidenced by elimination of individuals or populations sensitive to the pollutant. Differential effects on individual species within a community can occur through direct effects on sensitive species and through alteration of the relative competitive potential of species within the plant community.

Particulate emissions have their greatest impact on terrestrial ecosystems near large emission sources. Particulate matter in itself constitutes a problem only in those few areas where deposition rates are very high. Ecological modification may occur if the particles contain toxic elements, even though deposition rates are moderate. Solubility of particle constituents is critical, since water-insoluble elements are not mobile within the ecosystem. Most of the material deposited by wet and dry deposition on foliar surfaces in vegetated areas is transferred to the soil where accumulation occurs in the litter layer.

8.12 REFERENCES

Adepipe, N. O., R. E. Barrett, and D. P. Ormrod. Phototoxicity and growth response of ornamental bedding plants to ozone and sulfur dioxide. J. Am. Soc. Hortic. Sci. 97:341-345, 1972.

American Phytopathological Society. Glossary of air pollution terms and selected reference list. Phytopathol. News 8:5-8, 1974.

Anderson, A. J., D. R. Meyer, and F. K. Mayer. Heavy metal toxicities: levels of nickel, cobalt, and chromium in the soil and plants associated with visual symptoms and variation in growth of an oat corp. Aust. J. Agric. Res. 24:557-571, 1973.

Anderson, P. J. The effect of dust from cement mills on the setting of fruits. Plant World 17:57-68, 1914.

Ashenden, T. W. Growth reductions in cocksfoot (*Dactylis glomerata* L.) as a result of SO_2 pollution. Environ. Pollut. 15:161-166, 1978.

Ashenden, T. W. The effects of long-term exposures to SO_2 and NO_2 pollution on the growth of *Dactylis glomerata* L. and *Poa pratensis* L. Environ. Pollut. 18:249-258, 1979.

Ashenden, T. W., and T. A. Mansfield. Influence of wind speed on the sensitivity of ryegrass to SO_2. J. Exp. Bot. 28:729-735, 1977.

Babich, H., and G. Stotzky. Air pollution and microbial ecology. CRC Crit. Rev. Environ. Control 4:353-420, 1974.

Ballentyne, D. J. Sulfite inhibition of ATP formation in plant mitrochondria. Phytochemistry 12:1207-1209, 1973.

Barton, V. R., S. B. McLaughlin, and R. K. McConathy. The effects of SO_2 on components of least resistance to gas exchange. Environ. Pollut. 21:255-265, 1980.

Bazzaz, M. B., and Govindjee. Effects of lead chloride on chloroplast reactions. Environ. Lett. 6:175-191, 1974.

Bell, J. N. B., and W. S. Clough. Depression of yield in ryegrass exposed to sulfur dioxide. Nature (London) 241:47-49, 1973.

Bennett, J. H., and A. C. Hill. Inhibition of apparent phytosynthesis by air pollutants. J. Environ. Qual. 2:526-530, 1973a.

Bennett, J. H., and A. C. Hill. Absorption of gaseous air pollutants by a standardized plant canopy. J. Air Pollut. Control Assoc. 23:203-206, 1973b.

Bennett, J. H., A. C. Hill, A. Soleimani, and W. H. Edwards. Acute effects of combination of sulfur dioxide and nitrogen dioxide on plants. Environ. Pollut. 9:127-132, 1975.

Berigari, M. S., C. F. Jordan, and C. A. Feickert. The Effect of Sulfur Dioxide on Yield and Growth of Kidney Beans (*Phaseolus vulgaris* L.). Argonne Nat. Lab. Radiol. Environ. Res. Div. Annu. Rep. (Ecol.), January-December 1974. pp. 51-61.

Berry, C. R. Relative sensitivity of red, jack, and white pine seedlings to ozone and sulfur dioxide. Phytopathology 61:231-232, 1972.

Berry, C. R. Age of pine seedlings with primary needles affects sensitivity to ozone and sulfur dioxide. Phytopathology 64:207-209, 1974.

Billings, W. D. Plants and the Ecosystem. 3rd ed. Wadsworth Publishing Company, Inc. Belmont, PA., 1978. pp. 1-62.

Black, V. J., and M. H. Unsworth. Effects of low concentrations of sulphur dioxide on net photosynthesis and dark respiration of Vicia faba. J. Exp. Bot. 30:473-483, 1979.

Boertitz, S., H. G. Daessler, and E. Friedrich. Effect of metallurgical dust on agricultural plants. Tech. Umweltschutz 15:247-254, 1976.

Bonte, J., L. DeCormis, and P. Louguet. Effect of sulfur dioxide pollution on the degree of opening of Pelargonium x hortorum stomata. C.R. Seances Acad. Sci. Ser. D 280:2377-2380, 1975.

Botkin, D. B. The role of species interactions in the response of a forest ecosystem to environmental perturbation. In: Systems Analysis and Simulation in Ecology, vol. IV. B. C. Patten, ed., Academic Press, New York, NY, 1976. pp. 147-173.

Botkin, D. B., J. F. Janak, and J. R. Wallis. Some ecological consequences of a computer model of forest growth. J. Ecol. 60:849-872, 1972.

Bradford, G. R. Boron. In: Diagnostic Criteria for Plants and Soils. H. G. Chapman, ed., University of California, Berkeley, CA, 1966. pp. 33-61.

Brady, N. C. The Nature and Properties of Soils, 8th edition. MacMillan Publishing Co., New York, NY, 1974.

Brandt, C. S., and W. W. Heck. Effects of air pollutants on vegetation. In: Air Pollution, vol. I: Air Pollution and Its Effects. A. C. Stern, ed., Academic Press, New York, NY, 1968. pp. 401-443.

Brandt, C. J., and R. W. Rhoades. Effects of limestone dust accumulation on composition of a forest community. Environ. Pollut. 3:217-225, 1972.

Brandt, C. J., and R. W. Rhoades. Effects of limestone dust accumulation on lateral growth of forest trees. Environ. Pollut. 4:207-213, 1973.

Buron, A., and G. Cornic. Effect of sulfur dioxide on gas exchange in white mustard (Sinapsis alba). Bull. Soc. Vaudoise Sci. Nat. 71:451-461, 1973.

Carlson, R. W. Reduction in the photosynthetic rate of Acer Quercus and Fraxinus species caused by sulphur dioxide and ozone. Environ. Pollut. 18:159-170, 1979.

Chamberlain, A. C. The movement of particles in plant communities. In: Vegetation and the Atmosphere, vol. 1: Principles. J. L. Monteith, ed., Academic Press, London, England, 1975. pp. 155-203.

Chapman, H. D. Zinc. In: Diagnostic Criteria for Plants and Soils. H. D. Chapman, ed. University of California, Berkeley, CA, 1966. pp. 484-499.

Chiba, O., and V. Tanaka. The effect of sulphur dioxide on the development of pine needle blight caused by Rhizosphaera kalkhoffii Bubak (I). J. Jap. For. Soc. 50:135, 1968.

Chilgren, J. D. The response of prairie deer mice to a field SO_2 gradient. In: Proceedings of the 4th Joint Conference Sensing of Environmental Pollutants. American Chemical Society, Washington, DC, 1978. pp. 61-65.

Clapham, W. B. J. Natural Ecosystems. The Macmillan Company, New York, 1973. p. 43.

Clough, W. S. The deposition of particles on moss and grass surfaces. Atmos. Environ. 9:1113-1119, 1975.

Cogbill, C. V. The history and character of acid precipitation in eastern North America. In: Proceedings of the First International Symposium on Acid Precipitation and the Forest Ecosystem, U.S. Department of Agriculture and Ohio State University, Columbus, Ohio, May 12-15, 1975. L. S. Dochinger and R. A. Seliga, eds., USDA For. Serv. Gen. Tech. Rep. NE-23, U.S. Department of Agriculture, Upper Darby, PA, 1976. pp. 363-370.

Coleman, R. The importance of sulfur as a plant nutrient in world crop production. Soil Sci. 101:230-239, 1966.

Constantinidou, H. A., and T. T. Kozlowski. Effects of sulfur dioxide and ozone on Ulmus americana seedlings. I. Visible injury and growth. Can. J. Bot. 57:170-175, 1979a.

Constantinidou, H. A., and T. T. Kozlowski. Effects of sulfur dioxide and ozone on Ulmus americana seedlings. II. Carbohydrates, proteins, and lipids. Can. J. Bot. 57:176-184, 1979b.

Costonis, A. C. Injury to eastern white pine by sulfur dioxide and ozone alone and in mixtures. Eur. J. For. Pathol. 3:50-55, 1973.

Cotrufo, C., and C. R. Berry. Some effects of a soluble NPK fertilizer on sensitivity of eastern white pine to injury from SO_2 air pollution. For. Sci. 16:72-73, 1970.

Coughenour, M. B. Grasslands Sulfur Cycle and Ecosystem Response to Low-Level SO_2. Ph.D. Thesis, Colorado State University, Fort Collins, CO, 1978.

Coughtrey, P. J., C. H. Jones, M. H. Martin, and S. W. Shales. Litter accumulation in woodlands contaminated by Pb, Zn, Cd and Cu. Oecologia 39:51-60, 1979.

Cowling, D. W., and L. H. P. Jones. Sulphur deficiency on two forage plants in England. Sulphur Inst. J. 6:11, 1971.

Cowling, D. W., and D. R. Lockyer. The effect of SO_2 on Lolium perenne L. grown at different levels of sulfur and nitrogen nutrition. J. Exp. Bot. 29:257-265, 1978.

Cowling, D. W., L. H. P. Jones, and D. R. Lockyer. Increased yield through correction of sulfur deficiency in ryegrass exposed to sulfur dioxide. Nature (London) 243:479-480, 1973.

Cowling, D. W., and M. J. Koziol. Growth of ryegrass (Lolium perenne L.) exposed to SO_2. I. Effects on photosynthesis and respiration. J. Exp. Bot. 29:1029-1036, 1978.

Czaja, A. T. Zementstaubwirkungen auf pflanzen: Die enstehung der Zementkrusten [The effects of cement dust on plants: the formation of cement crusts.] Qual. Plant Mater. Veg. 8:201-238, 1961.

Daessler, H. G., J. Ranft, and K. H. Rehn. The susceptibility of woody plants exposed to fluorine compounds and sulfur dioxide. Flora (Jena) 161:289-302, 1972.

Daines, R. H. Sulfur dioxide and plant response. J. Occup. Med. 10: 516-534, 1968.

Darley, E. F. Studies on the effect of cement-kiln dust on vegetation. J. Air Pollut. Control Assoc. 16:145-150, 1966.

Davis, D. D. Sulfur Dioxide and Deciduous Plants. Air Pollution and Plants Series. PA. Coop. Ext. Ser. U. Ed. 3-17, 1972a.

Davis, D. D. Sulfur Dioxide and Evergreens. Air Pollution and Plants Series. PA. Coop. Ext. Serv. U. Ed. 3-526, 1972b.

Davis, D. D., and R. G. Wilhour. Susceptibility of woody plants to sulfur dioxide and photochemical oxidants: a literature review. EPA 600/3-76-102, U.S. Environmental Protection Agency, Corvallis, OR, September 1976.

Davis, J. B., and R. L. Barnes. Effects of soil-applied fluoride and lead on growth of loblolly pine and red maple. Environ. Pollut. 5:35-44, 1973.

Demeritt, M. E., W. M. Chang, J. D. Murphy, and H. D. Gerhold. Selection system for evaluating resistance of Scotch pine seedlings to ozone and sulfur dioxide. In: Proceedings of the 19th Northeastern For. Tree Improvement Conference, 1971. pp. 87-97.

Denison, W. C. Life in tall trees. Sci. Am. 228:75-80, 1973.

DeSloover, J., and F. LeBlanc. Mapping of atmospheric pollution on the basis of lichen sensitivity. In: Proceedings of the Symposium Recent Advances Tropical Ecology R. Misra and B. Gopal eds., The International Society for Tropical Ecology, Varanashi-5, India, 1968. pp.42-56.

Dochinger, L. S., and W. W. Heck. An ozone-sulfur dioxide synergism produces symptoms of chlorotic dwarf of eastern white pine. Phytopathology 59:399, 1969.

Dochinger, L. S., and K. F. Jensen. Effects of chronic and acute exposure to sulfur dioxide on the growth of hybrid poplar cuttings. Environ. Pollut. 9:219-229, 1975.

Dodd, J. L., W. K. Lauenroth, G. L. Thor and Coughenour. Effects of Chronic Low Levels SO_2 Exposure on Producers and Litter Dynamics. In: The Bioenvironmental Impact of a Coal-Fired Power Plant. Third Interim, Colstrip, Montana, December, 1977. E. M. Preston and T. L. Gullet, eds. EPA-600/3-79-044. U.S. Environmental Protection Agency, 1978. pp. 384-493.

Domes, W. Unterschiedlich CO_2-Abhangigkeit des Gasaustaushes beider Blattseiten von Zea Mays. [Different CO_2-sensitivities of the gas exchanges of the two leaf surfaces of Zea Mays.] Planta 98:186-189, 1971.

Donaubauer, E. Sekundärschäden in Österreichischen Rauchschadensgebieten. Schwierigkeiten der Diagnose and Bewertung. [Secondary damages in Austrian smoke damaged areas. Difficulties of diagnosis and assessment.] In: Materialy VI Miedzynardowej Konferencji, Polska Academia Nauk, Katowice, Poland. pp. 277-284. 1968.

Dreisinger, B. R. Sulfur dioxide levels and the effects of the gas on vegetation near Sudbury, Ontario. Presented at the 58th Annual Meeting, Air Pollution Control Association, Toronto, Ontario, 1965. Paper no. 65-121.

Dreisinger, B. R. The impact of sulphur dioxide pollution of crops and forests, Pollution and Our Environment, Conference Background Paper, vol. 1, Montreal, Canadian Council of Resource Ministers, Paper No. A4-2-1, 1967. p. 1-7

Dreisinger, B. R., and P. C. McGovern. Monitoring atmospheric sulfur dioxide and correlating its effects on crops and forests in the Sudbury area. In: Impact of Air Pollution on Vegetation, Air Pollution Control Association Specialty Conference. S. N. Linzon, ed., Ontario Department of Energy and Resource Management, Toronto, 1970. 122 pp.

Durran, D. R., M. J. Meldgin, M. Liu, T. Thoem, and D. Henderson. A study of long-range air pollution problems related to coal development in the northern Great Plains. Atmos. Environ. 13: 1021-1037, 1979.

Embleton, T. W. Magnesium. In: Diagnostic Criteria for Plants and Soils. H.D. Champman, ed., University of California Division of Agricultural Sciences, Riverside, CA, 1966. pp. 225-263.

Enderlein, H., and W. Kastner. What effect has a nutrient deficiency on the resistance of one-year-old conifers to SO_2? Arch. Forstwes. 16:431-435, 1976.

Evans, L. S., and P. R. Miller. Histological comparison of single and additive O_3 and SO_2 injuries to elongating ponderosa pine needles. Am. J. Bot. 62:416-421, 1975.

Faller, N. Effects of atmospheric SO_2 on plants. Sulfur Institute J. 6:5-7, 1970.

Faller, N. Plant nutrient sulfur-SO_2 versus SO_4. Sulfur Institute J. 7:5-6, 1971.

Fankhauser, N., C. Brunold, and K. H. Erismann. The influence of sublethal concentrations of sulfur dioxide on morphology, growth and production yield of the duckweed Lemna minor L. Oecologia 23:201-206, 1976.

Ferenbaugh, R. W. Effects of prolonged exposure of Oryzopsis hymenoides to SO_2. Water, Air, Soil Pollut. 10:27-31, 1978.

Fujiwara, T., T. Umezawa, and H. Ishikawa. Effects of mixed air pollutants on vegetation. I. Effects of combinations of SO_2, NO_2 and O_3. Cent. Inst. of Elec. Power. Res. Rep. 72007, Institute of Agricultural Electricity, 1973.

Garland, J. A. Dry and wet removal of sulfur from the atmosphere. Atmos. Eviron. 12:349-362, 1978.

Gilbert, O. L. An alkaline dust effect on epiphytic lichens. Lichenologist 8:173-178 1976.

Gerhold, H. D. Effect of air pollution on Pinus strobus L. and genetic resistance. EPA-600/3-77-002, U.S. Environmental Protection Agency, Corvallis, OR, January 1977.

Godzik, S., and H. F. Linskens. Concentration changes of free amino acids in primary bean leaves after continuous and interrupted SO_2 fumigation and recovery. Environ. Pollut. 7:25-38, 1974.

Granat, L., H. Rehe, and R. O. Hallberg. The global sulfur cycle. Ecol. Bull. 22:89-134, 1976.

Guderian, R. Reaktionen von Pflanzen-gemeinschaften des Feldfutterbaues auf Schwefeldioxideinwirkungen. [Responses of plant communities to air pollution.] Essen: Girardet-Verlag, Schriftenr. Landesanst. Immissions-Bodennutzungssch. d. Landes Nordrhein-Westfalen 4:80-100, 1966.

Guderian, R. Air Pollution: Phytotoxicity of Acidic Gases and its Significance in Air Pollution Control. Springer-Verlag, Berlin, W. Germany, 1977.

Guderian, R. Terrestrial Vegetation--Air Pollutant Interactions: Non-Gaseous Air Pollutants. Presented at the International Conference on Air Pollutants and Their Effects on the Terrestrial Ecosystem. Banff, Alberta, Canada, May 10-17, 1980.

Guderian R., and H. Stratmann. Freilandversuche zur Ermittlung von Schwefeldioxidwirkungen auf die Vegetation. III. Grenzwerte schädlicher SO_2-Immissionen für Obst-und Forstkulturen sowie für landwirtschaftliche and gärtnerische Pflanzenarten. Forschungber. [Field experiments to determine the effects of sulfur dioxide on vegetation. III Threshold values of harmful SO_2 pollution on fruit and forest culture as well as agricultural and horticultural species of plants.] Landes Nordrhein-Westfalen Nr. 1920. Westdeutscher Verlag, Cologne, 1968. 113 pp.

Haisman, D. R. The effect of sulfur dioxide on oxidizing enzyme systems in plant tissue. J. Sci. Food Agric. 25:803-810, 1974.

Halbwachs, G. Der Wasserhaushalt rauchgeschädigter Holzgewächse. [Water conservation in smoke damaged woody plants.] Allg. Forstz. 78:196-197, 1967.

Hallgren, J-E. Physiological and biochemical effects of sulfur dioxide on plants. In: Sulfur in the Environment, vol. 2. J. O. Nriagu, ed., John Wiley & Sons, New York, NY, 1978. pp. 163-210.

Hawksworth, D. L. Lichens as litmus for air pollution: a historical review. Int. J. Environ. Stud. 1:281-296, 1971.

Hawksworth, D. L., and F. Rose. Qualitative scale for estimating sulfur dioxide air pollution in England and Wales using epiphytic lichens. Nature (London) 227:145-148, 1970.

Heagle, A. S. Interactions between air pollutants and plant parasites. Annu. Rev. Phytopathol. 11:365-388, 1973.

Heagle, A. S., and J. W. Johnston. Variable responses of soybeans to mixtures of ozone and sulfur dioxide. J. Air Pollut. Control Assoc. 29:729-732, 1979.

Heagle, A. S., D. E. Body, and G. E. Neely. Injury and yield responses of soybean to chronic doses of ozone and sulfur dioxide in the field. Phytopathology 64:132-136, 1974.

Heck, W. W., and C. S. Brandt. Effects on vegetation: native, crops, forest. In: Air Pollution. Vol. II: The Effects of Air Pollution. A. C. Stern, ed., 3rd ed., Academic Press, New York, NY, 1977. pp. 157-229.

Heck, W. W., and J. A. Dunning. Response of oats to sulfur dioxide: interactions of growth temperature with exposure temperature or humidity. J. Air Pollut. Control Assoc. 28:241-246, 1978.

Heggestad, H. E., K. L. Tuthill, and R. N. Stewart. Differences among poinsettias in tolerance to sulfur dioxide. Horti. Sci. 8:337-338, 1973.

Heitschmidt, R. K., W. K. Lauenroth, and J. L. Dodd. Effects of controlled levels of sulfur dioxide on western wheatgrass in a southeastern Montana grassland. J. Appl. Ecol. 14:859-868, 1978.

Hill, A. C., S. Hill, C. Lamb, and T. W. Barrett. Sensitivity of native desert vegetation to SO_2 and to SO_2 and NO_2 combined. J. Air Pollut. Control Assoc. 24:153-157, 1974.

Hofstra, G., and D. P. Ormrod. Ozone and sulfur dioxide interaction in white bean and soybean. Can. J. Plant Sci. 57:1193-1198, 1977.

Horsman, D. C., and A. R. Wellburn. Synergistic effect of SO_2 and NO_2 polluted air upon enzyme activity in pea seedlings. Environ. Pollut. 8:123-133, 1975.

Horsman, D. C., and A. R. Wellburn. Appendix II. Guide to metabolic and biochemical effects of air pollutants on higher plants. *In*: Effects of Air Pollutants on Plants. T. A. Mansfield, ed., Cambridge University Press, Cambridge, England 1976. pp. 185-199.

Horsman, D. C., T. M. Roberts, and A. D. Bradshaw. Evolution of sulfur dioxide tolerance in perennial ryegrass. Nature (London) 276:493-494, 1978.

Hou, L-Y., A. C. Hill, and A. Soleimani. Influence of CO_2 on the effects of SO_2 and NO_2 on alfalfa. Environ. Pollut. 12:7-16, 1977.

Houston, D. B. Response of selected *Pinus strobus* L. clones to fumigation with sulfur dioxide and ozone. Can. J. For. Res. 4:65-68, 1974.

Houston, D. C., and G. R. Stairs. Genetic control of sulfur dioxide and ozone tolerance of eastern white pine. For. Sci. 19:267-271, 1973.

Husar, R. B., D. E. Patterson, J. D. Husar, N. V. Cillam and W. E. Wilson. Sulfur budget of a power plant plume. Atmos. Environ. 12:549-568, 1978.

Jacobson, J. S., and L. J. Colavito. The combined effect of sulfur dioxide and ozone on bean and tobacco plants. Environ. Exp. Bot. 16:277-285, 1976.

Jacobson, J. S., and A. C. Hill. Recognition of Air Pollution Injury to Vegetation: A Pictorial Atlas. Air Pollution Control Association, Pittsburgh, PA, 1970.

Jager, H. J., and H. Klein. Biochemical and physiological detection of sulfur dioxide injury to pea plants (*Pisum sativum*). J. Air Pollut. Control Assoc. 27:464-466, 1977.

Jancarik, V. Vyskyt drevokaznych hub v hourem poskozovani oblasti Krusnych hor. [Occurrences of wood-destroying fungi in the smoke-damaged area of the Erzgebirge.] Lesnictvi 34(7):677-692, 1961.

Jensen, K. F., and T. T. Kozlowski. Absorption and translocation of sulfur dioxide by seedlings of four forest tree species. J. Environ. Qual. 4:379-382, 1975.

Jennings, O. E. Smoke injury to shade trees. Natl. Shade Tree Conf. 10:44-48, 1934.

Jones, H. C. Sulfur oxides and particulate matter standards relative to ecological effects: differing viewpoints. *In*: Proceedings of the Proposed SO_x and Particulate Standard, Atlanta, GA, Sept. 16-18, 1980, Air Pollution Control Association, ed., 1981. pp. 84-94.

Jones, H. C., D. Weber, and D. Balsillie. Acceptable limits for air pollution dosages and vegetation effects: sulfur dioxide. Presented at the 67th Annual Meeting, Air Pollution Control Association, Denver, CO, 1974. Paper no. 74-225.

Jordan, M. Effects of zinc smelter emissions and fire on a chestnut oak woodland. Ecology 56:78-91, 1975.

Kamprath, E. J. Possible effects from sulfur in the atmosphere. Combustion 44:16-17, 1972.

Karnosky, D. F. Threshold levels for foliar injury to *Populus tremuloides* by sulfur dioxide and ozone. Can. J. For. Res. 6:166-169, 1976.

Karnosky, D. F. Evidence for genetic control of response to sulfur dioxide and ozone in *Populus tremuloides*. Can. J. For. Res. 7:437-440, 1977.

Katz, M. Sulfur dioxide in the atmosphere and its relation to plant life. Ind. Eng. Chem. 41:2450-2465, 1949.

Katz, M., and A. W. McCallum. The effect of sulfur dioxide on conifers. In: Proceedings of the U.S. Tech. Conf. Air Pollution, 1952. pp. 84-96.

Katz, M. Effect of contaminants other than sulfur dioxide on vegetation and animals. In: Pollution and Our Environment Conf., Background Paper A4-2-2, Vol. 1, Council Res. Ministers, Montreal, Can. 1967. pp. 1-18.

Keller, T. The effect of long duration, low SO_2 concentrations upon photosynthesis of conifers. In: Proceedings of the 4th International Clean Air Congress, Tokyo, Japan, 1977. pp. 81-83.

Keller, T. Wintertime atmospheric pollutants--do they effect the performance of deciduous trees in the ensuing growing season. Environ. Pollut. 16:243-247, 1978.

Keller, T. The effect of a continuous springtime fumigation with SO_2 and CO_2 uptake and structure of the annual ring in spruce. Can. J. For. Res. 10:1-6, 1980.

Kender, W. J., and F. H. F. G. Spierings. Effects of sulfur dioxide, ozone, and their interactions on "golden delicious" apple trees. Neth. J. Plant Pathol. 81:149-151, 1975.

Kickert, R. N., and P. R. Miller. Responses of ecological systems. In: Methodology for the Assessment of Air Pollution Effects on Vegetations. W. W. Heck, S. V. Krupa, and S. N. Linzon, eds., Air Pollution Control Association, Pittsburgh, Pa., 1979.

Kloke, A. Lead-zinc-cadmium enrichment in soils and plants. Staub-Rein. Luft 34:20-24, 1974.

Kodata, M., and T. Inoue. Invading path of sulfur dioxide into pine leaves as revealed by microradioautography. J. For. Soc. 54:207-208, 1972.

Krause, G. H. M., and H. Kaiser. Plant response to heavy metals and sulfur dioxide. Environ. Pollut. 12:63-71, 1977.

Krouse, H. R. Sulphur isotope abundance elucidate uptake of atmospheric sulphur emissions by vegetation. Nature (London) 265:45-46, 1977.

Krupa, S. V., B. I. Chevone, S. Fagerlie, F. Russo and D. F. Lang. 1976. Impact of air pollutants on terrestrial vegetation - a literature survey. State of Minnesota Environmental Quality Council.

Krupa, S. V., and R. J. Kohut. Impact of stark emissions from the NSP-SHERCO power plant on terrestrial vegetation. Annual Report, Northern States Power Co., Minneapolis, MN, 1976.

Kudela, M., and E. Novakova. Lesni skudci a skody zerí v lesích poškozovaných Kourum. [Insect and wild life damages in smoke-damaged forest stands.] Lesnictvi 35(6):493-502, 1962.

Labanauskas, C. K. Manganese. In: Diagnostic Criteria for Plants and Soils. H. D. Chapman, ed., University of California, Berkeley, CA, 1966. pp. 264-285.

Lacasse, N. L., and W. J. Moroz. Handbook of effects assessment; vegetation damage. CAES. Pennsylvania State University, University Park, PA, 1969.

Lagerwerff, J. V. Lead, mand cadmium as environmental contaminants. In: Micronutrients in Agriculture. J. J. Mortuedt, P. M. Giodana, and W. L. Lindsay, eds., Soil Science Society of America, Madison, WI, 1972. pp. 593-636.

Lauenroth, W. K., and J. E. Heasley. Impact of atmospheric sulfur deposition on grassland ecosystems. In: Atmospheric Sulfur Deposition: Environmental Impacts and Health Effects. D. S. Shriner, C. R. Richmond, and S. E. Lindberg, eds., Ann Arbor Science Publishers, Ann Arbor, Mi., 1980. pp. 417-430.

Laurence, J. A. Effects of air pollutants on plant pathogen interaction. In: Proceedings of the 71st Annual Meeting, Air Pollution Control Association, Houston, TX, 1978. pp. 3-24.

Laurence, J. A. Response of maize and wheat to sulfur dioxide. Plant Dis. Rep. 63:468-471, 1979.

Laurence, J. A., and L. G. Weinstein. Effects of sulfur dioxide on southern corn leaf blight maize and stem rust of wheat. Plant Dis. Reptr. 63:975-978, 1979.

Laurence, J. A., A. L. Alusio, L. H. Weinstein, and D. C. McCune. Effects of sulphur dioxide on southern bean mozaic and maize dwarf moziac. Environ. Pollut., 1981. (in press)

LeBlanc, F., and D. N. Rao. Effects of sulfur dioxide on lichen and moss transplants. Ecology 54:612-617, 1973.

LeBlanc, F., and D. N. Rao. Effects of air pollutants on lichens and bryophytes. In: Responses of Plants to Air Pollution. J. B. Mudd and T. T. Kozlowski, eds., Academic Press, New York, NY, 1975. pp. 237-272.

Lee, J. J., E. M. Preston, and R. A. Lewis. A system for the experimental evaluation of the ecological effects of sulfur dioxide. Proceedings of the 4th Joint Conference on Sensing of Environmental Pollutants, American Chemical Society, Washington, DC, 1978. pp. 49-53.

Legge, A. H. Primary Productivity, Sulfur Dioxide and the Forest Ecosystem: an overview of a Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, CA., P. Miller, ed. Pacific Southwest Forest and Range Experiment Station, Berkeley,CA, 1980. pp. 51-62.

Legge, A. H., C. W. Harver, P. F. Lester, D. R. Jaques, H. R. Krouse, J. Mayo, A. P. Hartgerink, R. G. Amundson, and R. B. Walker. Quantitative assessment of the impact of sulfur gas emissions on a forest ecosystem. Final report submitted to Whitecourt Environmental Study Group. Environmental Sciences Centre, Kananskis, The University of Calgary, Alberta, 1976.

Legge, A. H., D. R. Joques, G. W. Harvey, H. R. Krouse, H. M. Brown, E. C. Rhodes, M. Nosal, H. O. Schellhase, J. Mayo, A. P. Hartgerink, P. F. Lester, R. G. Amudson, and R. B. Walker. Sulphur gas emissions in the Boreal Forest: the West Whitecourt case study. Water, Air, and Soil Pollut. 15:77-85, 1981.

Leone, I. A., and E. Brennan. Modification of sulfur dioxide injury to tobacco and tomato by varying nitrogen and sulfur nutrition. J. Air Pollut. Control Assoc. 22:544-547, 1972.

Lerman, S. L., and E. F. Darley. Particles. In: Responses of Plants to Air Pollution. J. B. Mudd and T. T. Kozlowski, eds., Academic Press, New York, NY, 1975. pp. 141-158.

Liebig, G. F. Arsenic. In: Diagnostic Criteria for Plants and Soils. H. D. Chapman, ed., University of California, 1966. pp. 13-23.

Likens, G. E., F. H. Bormann, R. S. Pierce, J. S. Eaton, and N. M. Johnson. Biogeochemistry of a Forested Ecosystem. Springer-Verlag, New York, 1977.

Lindberg, S. E., A. C. Harriss, R. R. Turner, D. S. Shriner, and D. D. Huff. Mechanisms and rates of atmospheric deposition of selected trace elements and sulfate to a deciduous forest watershed. Pub. No. 1299, Oak Ridge National Laboratory, Oak Ridge, TN, 1979.

Linzon, S. N. The influence of smelter fumes on the growth of white pine in the Sudbury region. Canadian Department of Agriculture Publication, Ontario Dept. Lands Forests, 1958.

Linzon, S. N. Damage to eastern white pine by sulfur dioxide, semimature tissue needles blight and ozone. J. Air Pollut. Control Assoc. 16:140-144, 1966.

Linzon, S. N. Economic effects of sulphur dioxide on forest growth. J. Air Pollut. Control Assoc. 21:81-86, 1971.

Linzon, S. N. Some effects of particulate matter on vegetation. In: Ontario Proceedings of the 3rd International Clean Air Congress. Dusseldorf, West Germany, 1973. pp. A118-A120.

Linzon, S. N. Effects of sulfur oxides on vegetation. For. Chron. 48:182-186, 1972.

Linzon, S. N. Vegetation injury by airborne arsenic and sulphur dioxide emissions from gold smelters. Proceedings of the Fourth International Clean Air Congress, Tokyo, Japan, 1977.

Linzon, S. N. Effects of airborne sulfur pollutants on plants. In: Sulfur in the Environment: Part II, Ecological Impacts. J. O. Nriagu, ed., John Wiley & Sons. New York, NY, 1978. pp. 109-162.

Linzon, S. N. Acute and chronic effects of sulfur dioxide on natural vegetation. In: Proceedings of the Specialty Conference on the Proposed SO_x and Particulate Standard. Atlanta, GA, September 16-18, 1980. Air Pollution Control Association, ed., 1981. pp. 50-61.

Linzon, S. N. P. J. Temple and R. G. Pearson. Sulfur concentration in plant foliage and related effects. J. Air Pollution Control Association 29:520-525, 1979.

Little, P. A study of heavy metal contamination of leaf surfaces. Environ. Pollut. 5:159-172, 1973.

Little, P. Deposition of 2.75, 5.0 and 8.5mm particles to plant and soil surfaces. Environ. Pollut. 12:293-305, 1977.

Little, P., and R. D. Wiffen. Emission and deposition of petrol engine exhaust Pb-I. Deposition of exhaust Pb to plant and soil surfaces. Atmos. Environ. 11:437-447, 1977.

Majernik, O., and T. A. Mansfield. Direct effect of SO_2 pollution on the degree of opening of stomata. Nature (London) 227:377-378, 1970.

Malhotra, S. S., and A. A. Kahn. Effects of sulfur dioxide fumigation on lipid biosynthesis in pine needles. Phytochemistry 17:241-244, 1978.

Mandl, R. H., L. H. Weinstein, and M. Keveny. Effects of hydrogen fluoride and sulfur dioxide alone and in combination on several species of plants. Environ. Pollut. 9:133-143, 1975.

Manning, W. J. Effects of limestone dust on leaf condition, foliar disease incidence, and leaf surface microflora of native plants. Environ. Pollut. 2:69-76, 1971.

Mansfield. T. A., and O. Majernik. Can stomata play a part in protecting plants against air pollution? Environ. Pollut. 1:149-154, 1970.

Markowski, A., S. Grzesiak, and M. Schramel. Indexes of susceptibility of various species of cultivated plants to sulfur dioxide action. Bull. Acad. Pol. des Sci. 23(9):637-646, 1975.

Masaru, N., F. Syozo, and K. Saburo. Effects of exposure to various injurious gases on germination of lily pollen. Environ. Pollut. 11:181-187, 1976.

Materna, J., J. Jirgle, and J. Kucera. Vysledky merceni koncentraci lyclicniky siricitheo v. lesich krusnych hor. (Measurement results of sulfur dioxide concentrations in the Ore Mountain Forests.) Ochr. Ovzdusi. 6:84-93, 1969.

Matsushima, J., and R. F. Brewer. Influence of sulfur dioxide and hydrogen fluoride as a mix or reciprocal exposure on citrus growth and development. J. Air Pollut. Control Assoc. 22:710-713, 1972.

May, P. F., and A. M. Downes. Nutrient cycling in grazed pastures. I. A preliminary investigation of the use of [^{35}S] gypsum. Aust. J. Agric. Res. 19:531-543, 1968.

May, P. F., A. R. Till, and M. J. Cumming. Systems analysis of ^{35}sulfur kinetics in pastures grazed by sheep. J. Appl. Ecol. 9:25-49, 1972.

McLaughlin, S. B. SO_2, Vegetation effects and the air quality standard: limits of interpretation and application. In: Proceedings of the Proposed SO_x and Particulate Standard, Atlanta, GA, September 16-18, 1980, Air Pollution Control Association, ed., 1981, pp. 62-83.

McLaughlin, S. B., and N. T. Lee. Botanical Studies in the Vicinity of the Widows Creek Steam Plant. Review of Air Pollution Effects Studies, 1952-1972, and Results of 1973 Surveys. Internal Report I-EB-74-1, TVA, 1974.

McLaughlin, S. B., and D. S. Shriner. Allocation of resources to defense and repair. In: Plant Disease: An Advanced Treatise. J. G. Horsfall and E. B. Cowling, eds., Academic Press, New York, NY, 1980.

McLaughlin, S. B., and G. E. Taylor. Relative humidity: important modifier of pollutant uptake by plants. Science (Washington, DC) 211:167-169, 1981.

McLaughlin, S. B., V. J. Schorn, and H. C. Jones. A programmable exposure system for kinetic dose-response studies with air pollutant J. Air Pollut. Control Assoc. 26:132-135. 1976.

McLaughlin, S. B., D. S. Shriner, R. K. M'Conathy, and L. K. Mann. The effects of SO_2 dosage kinetics and exposure frequency on photosynthesis and transpiration of kidney beans (Phaseolus vulgaris L). Environ. Exp. Bot. 19:179-191, 1979.

Meidner, H., and T. A. Mansfield. Physiology of stomata. McGraw-Hill, Great Britian, 1968.

Menser, H. A., and H. E. Heggestad. Ozone and sulfur dioxide synergism: injury to tobacco plants. Science (Washington, DC) 153:424-425, 1966.

Meszaros, E., G. Varhelyi, and L. Haszpra. On the atmospheric sulfur budget over Europe. Atmos. Environ. 12:2273-2277, 1978.

Miller, P., and J. R. McBride. Effects of air pollutants on forest. In: Response of Plants to Air Pollution. J. B. Mudd and T. T. Kozlowski, eds., Academic Press, New York, NY, 1975. pp. 195-235.

Miller, J. E., H. J. Smith, P. G. Sprugel, and P. B. Xerikos. Yield response of field-grown soybeans to an acute SO_2 exposure. Radiol. Environ. Res. Div. Annu. Rep., Argonne National Laboratory, ANL-78-65, Part III, 1979.

Miller, J. E., D. G. Sprugel, R. N. Muller, H. J. Smith, and P. B. Xerikas. Open-air fumigation system for investigating sulfur dioxide effects on crops. Phytopathology 70: 1124-1128, 1980.

Moss, M. R. Biogeochemical cycles as integrative and spatial modes for the study of environmental pollution (the example of the sulphur cycle). Int. J. Environ. Stud. 9:209-216, 1976.

Mudd, J. B. Sulfur dioxide. In: Responses of Plants to Air Pollution. J. B. Mudd and T. T. Kozlowski, eds., Physiological Ecol. Mono. Series, Academic Press, Inc., New York, NY, 1975. pp. 9-22.

Murray, J. J., R. K. Howell, and A. C. Wilton. Differential response of 17 Poa pratensis cultivars to ozone and sulfur dioxide. Plan Dis. Rep. 59:852-854, 1975.

National Academy of Sciences. Series on Medical and Biological Effects of Environmental Pollutants. National Academy of Sciences, Washington, DC, 1973.

National Academy of Sciences. Principles for Evaluating Chemicals in the Environment. National Academy of Sciences, Washington, DC, 1975.

National Academy of Sciences. Sulfur oxides. Board on Toxicology and Environ. Health Hazards. U.S. Environmental Protection Agency Contract, Report No. 68-01-4655, 1978.

Neely, G. E., D. T. Tingey, and R. G. Wilhour. Effects of ozone and sulfur dioxide singly and in combination on yield, quality and N-fixation of alfalfa. Proceedings of the International Conference on Photochemical Oxidant Pollution and Its Control. EPA-600/3-77-001b, 1977. pp. 663-673.

Nielsen, D. G., L. E. Terrell, and T. C. Weidensaul. Phytotoxicity of ozone and sulfur dioxide to laboratory fumigated scotch pine. Plant Dis. Rep. 61:699-703, 1977.

Noggle, J. C., and H. C. Jones. Accumulation of Atmospheric Sulfur by Plants and Sulfur-supplying Capacity of Soils. EPA-600/7-79-109, U.S. Environmental Protection Agency, 1979.

Noland, T. L., and T. T. Kozlowski. Effect of SO_2 on stomatal aperature and sulfur uptake of woody angiosperm seedlings. Can. J. For. Res. 9:57-62, 1979.

O'Connor, J. A., D. G. Parbery, and W. Strauss. The effect of phytotoxic gases on native Australian plant species. Part I. Acute effects of sulfur dioxide. Environ. Pollut. 7:7-23, 1974.

Odum, E. P. Fundamentals of Ecology, 3rd ed. W. B. Saunders, Co., Philadelphia, PA, 1971.

O'Gara, P. J. Sulfur dioxide and fume problems and their solutions. In: 14th Semiannual Meeting of the American Institute of Chemical Engineers. J. Ind. Eng. Chem. 14:744, 1922.

Oshima, R. J. The impact of sulfur dioxide on vegetation: a sulfur dioxide-ozone response model. Final Report of the California Air Resources Board Agree. No. A6-162-30, 1978.

Pahlich, E. Allosterische Regulation der Aktivitat der Glutamat-dehydrogenase aus Erbsenkeimlingen durch das Substrat α-Ketoglutarsaure. [Allosteric regulation of the activity of glutamate dehydrogenase from pea seedlings by the substrate alpha-ketoglutarate.] Planta 100:222-227, 1971.

Pahlich, E. Uber den Hemm-Mechanismus mitochondriater Glutamat-Oxalacetat-transaminase in SO_2-begaster Erbsen. [Mechanism of inhibition of mitochondrial glutamate-oxalacetate-transaminase in sulfurdioxide fumigated peas.] Planta 110:267-278, 1973.

Pahlich, E. Effect of SO_2-pollution on cellular regulation. A general concept of the mode of action of gaseous air contamination. Atmos. Environ. 9:261-263, 1975.

Parish, S. B. The effect of cement dust on citrus trees. Plant World 13:288-291, 1910.

Pelz, E. Intersuchungen ueber die individuelle Rauchhaerte von Frehten [The individual smoke resistance of spruce.] Wiss. Z. Tech Univ., Dresden 11:3:595-600, 1962.

Preston, E. M. The ecological implications of chronic sulfur dioxide exposure for native grasslands. Presented at the 72nd Annual Meeting Air Pollution Control Association, Cincinnati, OH, 1979.

Preston, E. M., and R. A. Lewis. The Bioenvironmental Impact of a Coal Fired Power Plant, Third Interim Report, Colstrip, Montana, December, 1977. EPA-600/3-78-021, U.S. Environmental Protection Agency, 1978.

Rajput, C. B. S., D. P. Ormrod, and D. W. Evans. The resistance of strawberry to ozone and sulfur dioxide. Plant Dis. Rep. 61:221-225, 1977.

Rao, D. N., and F. LeBlanc. Effects of sulfur dioxide on the lichen alga with special reference to chlorophyll. Bryologist 69:69-75, 1965.

Reinert, R. A. Pollutant interactions and their effects on plants. Environ. Pollut. 9:115-116, 1975.

Reinert, R. A., and T. N. Gray. The response of radish to nitrogen dioxide, sulfur dioxide, and ozone alone and in combination. J. Environ. Qual., 10:240-243, 1981.

Reinert, R. A., and P. V. Nelson. Sensitivity and growth of five Elatior Begonia cultivars to SO_2 and O_3, alone and in combination. J. Am. Soc. Hort. Sci. 105:721-723, 1980.

Reinert, R. A., and D. E. Weber. Ozone and sulfur dioxide-induced changes in soybean growth. Phytopathology 70:914-916, 1980.

Reinert, R. A., A. S. Heagle, and W. W. Heck. Plant Response to Pollutant Combinations. In: Responses of Plants to Air Pollution. J. B. Mudd and T. T. Koslowski, eds., Academic Press Inc., New York, NY, 1975. pp. 159-177.

Rennie, P. J., and R. L. Halstead. The effects of sulfur on plants in Canada. In: Sulfur and its Inorganic Derivatives in the Canadian Environment. National Research Council Canada, NRCC No. 15015:69-179, 1977.

Reuther, W., and C. K. Labanauskas. Copper. In: Diagnostic Criteria for Plants and Soils. H. D. Chapman, ed., University of California, Berkeley, CA, 1966. pp. 157-179.

Ricks, G. R., and J. H. Williams. Effects of atmospheric pollution on deciduous woodland. Part 2: Effects of particulate matter upon stomatal diffusion resistance in leaves of *Quercus petraea* (Mattuschka) Leibl. Environ. Pollut. 6:87-109, 1974.

Rist, D. L., and D. D. Davis. The influence of exposure temperature and relative humidity on the response of pinto bean foliage to SO_2. Phytopathology 69:231-235, 1979.

Rosenberg, C. R., R. J. Hutnik, and D. D. Davis. Forest composition of varying distances from a coal-burning power plant. Environ. Pollut. 14:307-317, 1979.

Saunders, P. J. W. Effects of atmospheric pollution of leaf surface microflora. Pestic Sci. 4:589-594, 1973.

Schoenbeck, H. Beobachtungen zur Frage des Einflusses von industriellen Immissionen auf die Krankheitbereitschaft der Pflanze. [The effect of industrial pollution on the disease susceptibility of plants.] Berichte aus der Landestalt f. Bodennutzungsschutz des Landes Nordhein-Westfallen. pp 89-98, 1960.

Schoenbeck, H. Detection of heavy metal-containing air pollutants by selected plant indicators. Ver. Dtsch. Ing. 203:75-85, 1973.

Sehmel, G. A., and W. A. Hodgson. Predicted dry deposition velocities. *In*: Atmosphere-surface exchange of particulate and gaseous pollutants symposium. Conf-74092. Richland, WA, 1974. pp. 399-422.

Setterstrom, C., and P. W. Zimmerman. Factors influencing susceptibility of plants to sulfur dioxide injury. Contrib. Boyce Thompson Inst. 10:155-186, 1939.

Sharp, E. L. Atmospheric ions and germination of uredospores of *Puccinia striiformis*. Science (Washington, DC) 156:1359-1360, 1967.

Sharp, E. L. Relation of air ions to air pollution and some biological effects. Environ. Pollut. 3:227-239, 1972.

Sheridan, R. P. Impact of emissions from a coal-fired electricity generating facilties on N_2-fixing in lichens. Bryologist 82:54-58, 1979.

Shinn, J. H. A critical survey of measurements of foliar deposition of airborne sulfates and nitrates. Presented at the 71st Annual Meeting, Air Pollution Control Association, Houston, TX, June 25-30, 1978. Paper No. 78-7.2.

Shinn, J. H., and S. Lynn. Do manmade sources affect the sulfur cycle of northeastern states? Environ. Sci. Technol. 13:1062-1067, 1979.

Showman, R. E. Lichens as indicators of air quality around a coal-fired power generating plant. Bryologist 78:1-6, 1975.

Shriner, D. S., and G. S. Henderson. Sulfur distribution and cycling in a deciduous forest watershed. J. Environ. Qual. 7:392-397, 1978.

Shugart, H. H., and D. C. West. Development of an Appalachian deciduous forest succession model and its application to assessment of the impact of the Chestnut Blight. J. Environ. Manag. 5:161-179, 1977.

Sij; J. W., E. T. Kanemasu, and S. M. Goltz. Some preliminary results of sulfur dioxide effects on photosynthesis and yield in field-grown wheat. Trans Kans. Acad. Sci. 76:199-207, 1974.

Smith, H. J., and D. D. Davis. The influence of needle age on sensitivity of Scotch pine to acute doses of SO_2. Plant Dis. Rep. 61:870-874, 1977.

Smith, H. J., and D. D. Davis. Susceptibility of conifer cotyledons and primary needles to acute doses of sulfur dioxide. Hortic. Sci. 13:703-704, 1978.

Smith, R. L. Ecology and Field Biology, 3rd ed., Harper and Row, New York, NY, 1980. pp. 11-199.

Smith, W. H. Metal contamination of urban woody plants. Environ. Sci. Technol. 7:631-636, 1973.

Spedding, D. J. Uptake of sulfur dioxide by barley leaves at low sulfur dioxide concentrations. Nature (London) 224:1229-1231, 1969.

Spinka, J. Effects of polluted air on fruit trees and legumes. Ziva 19:13-15, 1971.

Sprugel, D. G., J. E. Miller, R. N. Muller, H. J. Smith, and P. B. Xerikos. Sulfur dioxide effects on yield and seed quality in field-grown soybeans. Phytopathology 70:1129-33, 1980.

Stewart, D., M. Treshow, and F. M. Harner. Sulfur dioxide and hydrogen fluoride emissions in regard to phytopathology. Can. J. Bot. 51:983-988, 1973.

Sulphur Institute. Crop Responses to Sulphur in North America, vol. 18. The Sulphur Institute, Washington, DC, 1971.

Sulphur Institute. Sulphur in Agriculture Series, vol. 3. J. S. Platu, ed., The Sulphur Institute, Washington, DC, 1979.

Temple, P. J. Dose-response of urban trees to sulfur dioxide. J. Air Pollut. Control Assoc. 22:271-274, 1972.

Terman, G. L. Atmospheric sulfur - the agronomic aspects. Tech. Bull. no. 23. The Sulfur Institute. Washington, DC, 1978. 15 p.

Thomas, M. D. Gas damage to plants. Ann. Rev. Plant Physiol. 2:293-322, 1951.

Thomas, M. D., and R. H. Hendricks. Effect of air pollution on plants. In: Air Pollution Handbook. P. L. Magill et al. eds., New York, NY, 1956. pp. 9:1-44.

Thomas, M. D., and G. R. Hill, Jr. Absorption of sulfur dioxide by alfalfa and its relation to leaf injury. Plant Physiol. 10:291-307, 1935.

Thomas, M. D., R. H. Hendricks, T. R. Collier, and G. R. Hill. The utilization of sulphate and sulphur dioxide for the sulphur nutrition of alfalfa. Plant Physiol. 18:345-371, 1943.

Thompson, L. K., S. S. Sidhu, and B. A. Roberts. Fluoride accumulations in soil and vegetation in the vicinity of a phosphorus plant. Environ. Pollut. 18:221-234, 1979.

Thompson, R. C., and G. Katz. Effects of continuous H_2S fumigation on crop and forest plants. Environ. Sci. and Tech. 12:550-553, 1978.

Tingey, D. T., and R. A. Reinert. Effect of ozone and sulfur dioxide singly and in combination on plant growth. Environ. Pollut. 9:117-125, 1975.

Tingey, D. T., W. W. Heck, and R. A. Reinert. Effect of low concentrations of ozone and sulfur dioxide on foliage, growth, and yield of radish. J. Am. Soc. Horti. Sci. 96:369-371, 1971b.

Tingey, D. T., R. A. Reinert, J. A. Dunning, and W. W. Heck. Vegetation injury from the interaction of nitrogen dioxide and sulfur dioxide. Phytopathology 61:1506-1511, 1971a.

Tingey, D. T., R. A. Reinert, J. A. Dunning, and W. E. Heck. Foliar injury responses of eleven plant species to ozone/ sulfur dioxide mixtures. Atmos. Environ. 7:201-208, 1973a.

Tingey, D. T., R. A. Reinert, C. Wickliff, and W. W. Heck. Chronic ozone or sulfur dioxide exposures, or both, affect the early vegetative growth of soybean. Can. J. Plant Sci. 53:875-879, 1973b.

Treshow, M. Environment and Plant Response. McGraw-Hill Book Co., New York, NY, 1970.

Tyler, G. Heavy metals pollute nature, may reduce productivity. Ambio 1:53-59, 1972.

Ulrich, A., and K. Ohki. Potassium. In: Diagnostic Criteria for Plants and Soils. H. D. Chapman, ed., University of California, Berkeley, CA, 1966. pp. 157-179.

Ulrich, A., M. A. Tabatabai, K. Ohki and C. M. Johnson. Sulfur content of alfalfa in relation to growth in filtered and unfiltered air. Plant Soil 26:235-252, 1967.

Unzicker, H. J., H. J. Jager, and L. Steubing. Influence of SO_2 on the vitamin content of plants. Angew. Bot. 49:131-139, 1975.

U.S. Department of Health, Education, and Welfare. Air Quality Criteria for Particulate Matter. Public Health Service, EHS, National Air Pollution Control Administration, Washington, DC 1969.

U.S. Environmental Protection Agency. Mount Storm, West Virginia -Gorman, Maryland, and Luke, Maryland - Keyser, West Virginia Air Pollution Abatement Activity, Research Triangle Park, N.C. 1971.

U.S. Environmental Protection Agency. Effects of Sulfur Oxides in the Atmosphere on Vegetation; revised chapter 5 for Air Quality Criteria for Sulfur Oxides. EPA-R3-73-030, National Environmental Research Center, U.S. Environmental Protection Agency, Raleigh, NC, 1973.

U.S. Environmental Protection Agency. The Bioenvironmental Impact of Fine Particulates: A Critical Review and Summary. USEPA/CERL Report, 1975. Mimeo, 49 pp.

U.S. Environmental Protection Agency. Diagnosing Vegetation Injury Caused by Air Pollution. D. R. Hicks, ed., EPA Contract Pub. No. 68-02-1344, Research Triangle Park, N.C. 1976.

U.S. Environmental Protection Agency. Airborne Particles. Environmental Health Effects Series, Report No. EPA-600/1-77-053, 1977.

U.S. Environmental Protection Agency. Office of Air Quality Planning and Standards. National Air Quality, Monitoring and Emissions Trends Report, 1977, EPA-450/2-78-052, U.S. Environmental Protection Agency, Research Triangle Park, NC, 1978b.

U.S. Environmental Protection Agency. Emission of Sulfur-Bearing Compounds from Motor Vehicle and Aircraft Engines. A Report to Congress. EPA-600/9-78-028, U.S. Environmental Protection Agency, Washington, D.C. August 1978a.

U.S. Environmental Protection Agency. Air Quality Criteria for Oxides of Nitrogen Draft Final: EPA-600/8-82-026, U.S. Environmental Protection Agency, Research Triangle Park, NC, September 1982.

Van Haut, H., and H. Stratmann. Farbtafelatlas uber Schwefeldioxidwirkugen and Pflanzen. (Color-Plate Atlas of the Effects of Sulfur Dioxide on Plants.) Verlag W. Girardet, Essen, West Germany, 1970.

Vanselow, A. P. Nickel. In: Diagnostic Criteria for Plants and Soil. H. D. Chapman, ed., University of California, Berkeley, CA, 1966. pp. 302-309.

Vins, B., and R. Mrkva. The diameter increment losses of pine stands as a result of injurious emissions. Acta Univ. Ser. C Agric.-Brno 42:25-46, 1973.

Vogl, M., S. Bortitz, and H. Polster. Physiological and biochemical contributions to research of fume damage. 6th report. Definitions of degrees of damage and forms of resistance against the SO_2 component of injurious fumes. Biol. Zentralb. 84:763-777, 1965.

Wainwright, M. Microbial S-oxidation in soils exposed to heavy atmospheric pollution. Soil Biol. Biochem. 11:95-98, 1979.

West, D. C., S. B. McLaughlin, and H. H. Shugart. Simulated forest response to chronic air pollution stress. J. Environ. Qual. 9:43-49, 1980.

White, K. L., A. C. Hill, and J. H. Bennett. Synergistic inhibition of apparent photosynthesis rate of alfalfa by combinations of sulfur dioxide and nitrogen dioxide. Environ. Sci. Technol. 8:574-576, 1974.

Wilhour, R. G., A. Neely, D. Weber and L. Grothaus. The response of selected small grains and range grasses, and alfalfa to SO_2. In: Bioenvironmental Impact of a Coal-fired Power Plant: An Interim Report. E. M. Preston and T. L. Gallett, eds., EPA-600/3-79-044, U. S. Environmental Protection Agency, Corvallis Environmental Res. Lab., Corvallis, OR, December 1978, pp. 592-609.

Winner, W. E., and J. D. Bewley. Terrestrial mosses as bioindicators of SO_2 pollution stress. Synecological analysis and the index of atmospheric purity. Oecologia 35:221-230, 1978a.

Winner, W. E., and J. D. Bewley. Contrasts between bryophyte and vascular plant synecological responses in an SO_2-stressed white spruce association in central Alberta. Oecologia 33:311-325, 1978b.

Winner, W. E. and H. A. Mooney. Ecology of SO_2 resistance: II Photosynthetic changes of shrubs in relation to SO_2 absorption and stomatal behavior. Oecologia 44:296-302, 1980.

Winner, W. E., J. D. Bewley, H. R. Krouse, and H. M. Brown. Stable sulfur isotope analysis of SO_2 pollution impact on vegetation. Oecologia 36:351-361, 1978.

Woodwell, G. Effects of pollution on structure and physiology of ecosystems. Science (Washington, DC) 168:429-433, 1970.

Wu, L., and A. D. Bradshaw. Aerial pollution and the rapid evolution of copper tolerance. Nature (London) 238:167-169, 1972.

Yopp, J. H., W. E. Schmid, and R. W. Holst. Determination of maximum permissible levels of selected chemicals that exert toxic effects on plants of economic importance in Illinois. Illinois Institute for Environmental Quality, 1974.

Zahn, R., Wirkungen von Schwefeldioxyd auf die Vegetation, Ergebnisse aus Begasungsversuchen. [Effects of sulfur-dioxide on vegetation: results of gas exposure experiments.] Staub Reinholt Luft. 21:56-60, 1961.

Zahn, R. Uber Einfluss verschiedener Umweltfaktoren auf die Pflanzenempfindlichkeit gegenüberSchwefeldioxyd. [Effects of various environmental factors on plant sensitivity to sulfur dioxide.] Z. Pflanzenkr. Pflanzenschutz 70:81-95, 1963a.

Zahn, R. Untersuchungen über die Bedeutung Kontinuierlicher and intermittierender Schwefeldioxideinwirkung fur die Pflanzenreaktion. (Investigations on plant reaction to continuous and/or intermittent sulfur dioxide exposure.) Staub Reinholt Luft. 23:343-352, 1963b.

Zahn, R. The effect on plants of a combination of subacute and toxic sulfur dioxide doses. Staub 30:20-23, 1970.

Ziegler, I. The effect of SO_3^{2-} on the ability of ribulose-1,5-diphosphate carboxylase in insulated spinach chloroplasts. Planta 103:155-163, 1972.

Ziegler, I. The effect of air polluting gases on plant metabolism. Environ. Qual. Saf. 2:182-208, 1975.

Zimmerman, P. W., and W. Crocker. Toxicity of air containing sulfur dioxide gas. Contrib. Boyce Thompson Inst. 6:455-470, 1934.

APPENDIX 8A

TABLE 8A-1. SUMMARY OF STUDIES REPORTING RESULTS OF SO_2 EXPOSURE UNDER FIELD CONDITIONS AND/OR CHAMBERS OVER PLANTS FOR AGRONOMIC CROPS

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c]		Species effect[d]	Caveat[e]	Reference
				Foliage	Yield			
100 (0.04)	3 hr for 8 exp. growing season	F/CC	Wheat			No effect on apparent photosynthesis, no effect on the avg. head length or no. of grains/head.		Sij et al., 1974
50 (0.02) 130 (0.05) 260 (0.10) (Geom. means = 58; 100; 178 μg/m³ or 0.02; 0.04; 0.07 ppm, respectively)	Growing season	F-ZAP	Western wheat grass, Prairie June grass			S content increased with increasing SO_2 conc.; digestible dry matter was decreased by 2 years of treatment; crude protein content in winter wheat decreased significantly.		Dodd et al., 1978
80-260 (0.03-0.10)	72 hr/wk for growing season	F/CC	Barley Durum wheat Spring wheat			No effect on yield		Wilhour et al., 1978
240 (0.09)	Mean conc. 4.2h/18 fumigations from 19 July-27 August	F-ZAP	Soybean cv. Wells	X	X	6.4% yield reduction	Sg[f] of fumigation conc. ranged 44-45% of x.	Sprugel et al., 1980 and Miller et al., 1980
260 (0.10)				X	X	5.2% yield reduction		
500 (0.19)				X	X	12.2% yield reduction		
660 (0.25)				X	X	19.2% yield reduction		
940 (0.36)				X	X	15.9% yield reduction		
260 (0.10)	6 hr/d 43 d 92 d 133 d	F/CC	Soybean cv. Dare		X	No significant effect on foliar injury, fresh wt. seeds/plant or wt. of seeds/plant. 92nd day defoliation was 12% > control; 135th day seed wt. only 1% < control.		Heagle et al., 1974
310 (0.12)	Mean conc. 4.7 hr/24 fumigations 13 July to 29 August	F-ZAP	Soybean cv. Wells	X	X	12.3% yield reduction	Sg[f] of fumigation conc. ranged 41-64% of x,	Sprugel et al., 1980 and Miller et al., 1980
790 (0.30)				X	X	20.5% yield reduction		
2070 (0.79)				X	X	45.3% yield reduction		

TABLE 8A-1. (continued)

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
390 (0.15)	72 hr/wk for growing season	F/CC	Barley Durum wheat Spring wheat		X X	44% lower yield in Barley (N.S.) 42% lower yield in Durum wheat (N.S.) No effect on Spring wheat		Wilhour et al, 1978
630 (0.25) 1050 (0.40) 2100 (0.80) 3140 (1.20)	Once every week (3 hr) to once in 5 wks (3 hr)	F/CC	Alfalfa Barley Durum wheat Spring wheat			No effect on yield No effect on yield No effect on yield No effect on yield		Wilhour et al., 1978
1180 (0.45)	3 hr for 7 exp. growing season	F/CC	Wheat			No accumulative effect on yield, no effect on avg. head length or no. grains/head		Sij et al., 1974
2100-5240 (0.80-2.00)	4 hr 20 min	F-ZAP	Soybean cv. Wells		X X X	4.5% lower yield at 3760 µg/m³ (1.4 ppm) 11% lower yield at 4450 µg/m³ (1.7 ppm) 15% lower yield at 5240 µg/m³ (2.0 ppm)	Pollutant avg.; no est. on range of exposure doses	Miller et al., 1979

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[a] Table arranged by increasing SO_2 concentration as first order and exposure time as second order divisions. Treatments within a single study that did not induce specifically different effects are listed along with the lowest SO_2 concentration that induced said effect.

[b] F/CC = field, closed chambers; F/OT = field, open top chambers, F-ZAP = field, zonal air pollution system.

[c] X indicates study found foliage and/or yield effects.

[d] Most prominent or significant effect reported.

[e] Caveats for consideration about proper study design and interpretations.

[f] Sg = Standard geometric deviation.

N.S. = Results were not significant at 95 percent level of confidence.

Note: 1 ppm SO_2 = 2620 µg/m³.

TABLE 8A-2. SUMMARY OF STUDIES REPORTING RESULTS OF SO_2 EXPOSURE UNDER LABORATORY CONDITIONS FOR AGRONOMIC AND HORTICULTURAL CROPS

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
92 (0.035) 465 (0.175)	8 hr	EC/SD	Broadbean		--	Depressed net photosynthesis	--	Black and Unsworth, 1979
130 (0.05)	5 hr/d; 5 d/wk for 4 wk	EC/SD	Alfalfa		X X	26% less foliage dry wt. at final harvest 49% less root dry wt. at final harvest		Tingey and Reinert, 1975
			Tobacco, Bel W3		X	22% less leaf dry wt. at final harvest	Sensitive plant	
			Tobacco, Burley 21			No effect		
130 (0.05) 260 (0.10) 660 (0.25)	4 hr	EC/SD	Oats Radish Soybean Tobacco		--	No foliar injury	--	Tingey et al., 1971a
130 (0.05) 520 (0.20)	8 hr/d 5 d/wk for 18 d	EC/SD	Soybean		--	No effect on top fresh or dry wt., root fresh or dry wt.; plant height, shoot/root fresh or dry wt. ratio	--	Tingey et al., 1973b
178 (0.068)	103.5 hr/wk for 20 wks	GC	Cocksfoot Meadowgrass		X X	40% less total dry wt. 20% less total dry wt.	--	Ashenden, 1979
180-1390 (0.07-0.53)	24 hr/d 9-20 d	GC	Tobacco		X	Increased dry wt. yield up to 1390 µg/m³ (0.53 ppm) (44% > control)	No monitoring methods presented; Low S in soil medium	Faller, 1970
			Sunflower		X	Greatest dry wt. yield: 44% > control at 920 µg/m³ (0.35 ppm); 27% > control at 1390 µg/m³ (0.53 ppm).		
			Corn		X	Greatest dry wt. yield: 24% > control at 450 µg/m³ (0.17 ppm); 7% > control at 1390 µg/m³ (0.53 ppm).		

TABLE 8A-2 (continued)

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
260 (0.10)	18 d	GC	Pea		X	3% less fresh wt. shoot 5% less dry wt. shoot 4% less total nitrogen 30% less H⁺ (buffer capacity) 10% more glutamate dehydrogenase activity 110% more inorganic sulfur content	--	Jager and Klein, 1977
290 (0.11)	24 hr/day 4 wk	GC	Italian ryegrass	X		No difference from control at low wind; 17%-40% less total dry wt. at high wind	--	Ashenden and Mansfield, 1977
325-2620 (0.125-1.0)	1 and 3 hr	EC/SD	Oats Radish Sweet pea Swiss chard		--	No foliar injury at concentrations $\leq$ 0.50 ppm; 2% maximum foliar injury experienced at all doses	--	Bennett et al., 1975
390 (0.15)	18 d	GC	Pea	X		3% less fresh wt. of shoot 8% less dry wt. of shoot 7% less total nitrogen 35% less H⁺ (buffer capacity) 32% more glutamate dehydrogenase activity 140% more inorganic sulfur content	Water culture	Jager and Klein, 1977
390-790 (0.15-0.30)	24 hr/d 7 d	EC/SD	Barley Bean Corn	X X X		Severe foliar injury No injury Severe foliar injury	--	Mandl et al., 1975
460 (0.175)	2 hr	EC/SD	Broadbean	--		dec. photosynthetic rate, dec. stomatal resistance if RH > 40%, inc. stomatal resistance if RH < 40%	--	Black and Unsworth, 1979

TABLE 8A-2 (continued)

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
520-790 (0.20-0.30)	2 hr	GC	Alfalfa Barley			Threshold dose for inhibition of photosynthesis, reversible effect		Bennett and Hill, 1973a
520 (0.20)	30, 70, 100 hr	EC/SD	Wheat	X	X	Trend of increased dry wt. for 19 of 21 exposures; small amount foliar injury	Trend, not significant from control	Laurence, 1979
520 (0.20)	15 d	GC	Tomato	X		Threshold dose for initial symptom of tissue death, dec. or no change in vitamin B_1, B_6, and nicotinic acid content		Unzicker et al., 1975
520 (0.20)	Continuous to maturation	EC/SD	Kidney bean		X	≤ 15% of total yield; no change in protein content		Berigari et al., 1974
660 (0.25)	4 hr	EC/SD	Broccoli Tobacco, Bel B Alfalfa Onion Soybean Lima bean Bromegrass Cabbage Radish Spinach Tomato	X X		6% leaf injury 1% leaf injury No effects No effects No effects No effects No effects No effects No effects No effects No effects		Tingey et al., 1973a
660 (0.25)	10 d	GC	Pea		X	32% less fresh wt. of shoot 26% less dry wt. of shoot 24% less total nitrogen 42% less H^+ (buffer capacity) 80% more glutamate dehydrogenase activity 150% more inorganic sulfur content	Water culture	Jager and Klein, 1977

TABLE 8A-2 (continued)

Conc.[a] μg/m^3 (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
2620 (1.00)	2 hr	EC	Begonia			No effect	--	Adedipe et al., 1972
			Petunia		X	30% less flower #'s; 19% less shoot wt.		
			Coleus		X	27% less flower #'s; 19% less shoot wt.		
			Snapdragon		X	14% less flower #'s; 16% less shoot wt.		
2620 (1.00)	4 hr	EC/SD	Broccoli	X		38% leaf injury	--	Tingey et al., 1973a
			Bromegrass	X		65% leaf injury		
			Cabbage	X		70% leaf injury		
			Lima bean	X		25% leaf injury		
			Radish	X		46% leaf injury		
			Spinach	X		49% leaf injury		
			Tomato	X		33% leaf injury		
2620 (1.00)	6 hr/d for 3 d	EC/SD	Strawberry	X		No effect on growth and development; necrotic lesions, lower leaf surface	--	Rajput et al., 1977
2620 (1.00)	1.5 hr	EC/SD	Soybean	X	X	9% less shoot fresh wt., 4% leaf injury	Short-term growth response only	Heagle and Johnston, 1979
	3 hr				X	21-29% less shoot fresh wt.		
3930 (1.50)	.75-3 hr	EC/SD	Soybean	X	X	24-94% less shoot fresh wt., 63-93% foliar injury	Short-term response only	Heagle and Johnston, 1979
3930 (1.50)	3 hr	EC/SD	Alfalfa	X		Leaf necrosis at 315 ppm CO_2 was 2.5x that induced under 645 ppm CO_2	--	Hou et al., 1977
3930 (2.00)	2 hr	EC	Begonia		X	14% fewer flowers; 22% less shoot wt.	--	Adedpie et al., 1972
			Petunia		X	32% fewer flowers; 24% less shoot wt.		
			Coleus		X	30% fewer flowers; 20% less shoot wt.		
			Snapdragon		X	15% fewer flowers; 15% less shoot wt.		

TABLE 8A-2 (continued)

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
660 (0.25)	4 hr, 3 times/wk 11 wk	EC/SD	Soybean		X	No foliar injury; significant dec. plant ht. at 5,7,9,11 wks; significant dec. shoot dry wt. at 7,11 wks; significant dec. root dry wt. at 9,11 wks; significant dec. total dry wt. at 11 wks	Grown in silica sand with Hoagland's solution as nutrient source	Reinert and Weber, 1980
790 (0.30)	5 hr/d 6 d/wk 12 d	EC/SD	Barley	X	X	11% foliar injury; 38% dec. dry wt. shoot	Monitoring system explained in unavailable publication	Markowski et al., 1975
			Bean	X	X	<1% foliar injury; 25% dec. dry wt. shoot		
			Sunflower	X	X	5% foliar injury; 41% dec. dry wt. shoot		
	26 d		Barley	X	X	21% foliar injury; 25% dec. dry wt. shoot		
			Bean	X	X	2% foliar injury; 15% dec. dry wt. shoot		
			Sunflower	X	X	16% foliar injury; 29% dec. dry wt. shoot		
920 (0.35)	1 hr	EC/SD	Alfalfa		--	8% decrease in apparent photosynthesis	--	White et al., 1974
1050 (0.40) 1310 (0.50) 1570 (0.60)	4 hr	EC/SD	Tomato	--		increased accumulation of total and soluble S content	--	Bennett and Hill, 1973a
1050 (0.40)	6 hr	EC/SD	Apples	X		2% leaf injury	--	Kender and Spierings, 1975
2620 (1.00)	2 hr	GC	Barley	X		Threshold dose for foliar necrosis; 30-60% decrease in net photosynthesis	--	Bennett and Hill, 1973a
2620 (1.00)	3 hr	GC	Poinsettia 8 cv.'s	X		Foliar injury 1 cultivar	--	Heggestad et al., 1973

TABLE 8A-2 (continued)

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
5240 (2.00)	3 hr	GC	Poinsettia 8 cv's	X		Foliar injury 2 cultivars	--	Heggestad et al., 1973
6550 (2.50)	6 hr	EC/SD	Apples	X	X	increased foliar injury; 62% more leaf abscission; 19% less shoot growth	--	Kender and Spierings, 1975
7860 (3.00)	1 hr	GD	Poinsettia 8 cv's	X		Foliar injury, 5 cv's	--	Heggestad et al., 1973
	2 hr	GC	Poinsettia 9 cv's	X		Foliar injury, 7 cv's		
	3 hr	GC	Poinsettia 8 cv's	X		Foliar injury, 8 cv's		
10480 (4.00)	2 hr	EC	Begonia	X	X	27% fewer flowers; 33% less shoot wt.; severe necrosis	--	Adedipe et al., 1972
			Petunia	X	X	42% fewer flowers; 32% less shoot wt.; slight injury		
			Coleus		X	30% fewer flowers; 21% less shoot wt.; no foliar injury		
			Snapdragon	X	X	20% fewer flowers; 19% less shoot wt.; slight injury		
			Marigolds	X		Slight injury		
			Celosia, Salvia	X		Slight injury		
			Impatiens	X		Slight injury		
1050 (0.40)	30, 78, 100 hr	EC/SD	Wheat 7 cv's	X		No effect on yield, small amount foliar injury	--	Laurence, 1979
1310 (0.50)	1.5 hr	EC/SD	Soybean	X	X	7% decrease in shoot fresh wt.; trace foliar injury	Short-term growth response only	Heagle and Johnston, 1979
1310 (0.50)	100 hr	EC/SD	Corn	--		Minimal foliar injury, no effect on dry mass	--	Laurence, 1979

TABLE 8A-2 (continued)

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
1570 (0.60)	6 hr	EC/SD	Apples	X		7.3% more foliage injury; 5% more leaf abscission	--	Kender and Spierings, 1975
1570 (0.60)	30, 78, 100 hr	EC/SD	Wheat 7 cv's	X	X	Trend of decreased dry wt. in 17 of the 21 exposures, small amount of foliar injury	Trend not significant from control	Laurence, 1979
1970 (0.75)	3 hr	EC/SD	Alfalfa	X		No injury developed	--	Hou et al., 1977
2100 (0.80)	2 hr	GC	Alfalfa	X		Threshold dose for foliar necrosis; 25-50% decrease in net photosynthesis	--	Bennett and Hill, 1973b

[a] Table arranged by increasing SO_2 concentration as first order and exposure time as second order division. Doses within a single study that did not induce specifically different effects are listed along with the lowest SO_2 concentration that induced said effect.

[b] GC = Growth chambers, EC = Exposure chambers, EC/SD = Exposure chamber, special design.

[c] X indicates study found foliage and/or yield effects.

[d] Most prominent or significant effect reported.

[e] Caveats for consideration about proper study design and interpretation.

[f] The symbol ">" means greater than; "<" means less than.

Note: 1 ppm SO_2 = 2620 ug/m³.

TABLE 8A-3. SUMMARY OF STUDIES REPORTING RESULTS OF SO_2 EXPOSURE UNDER LABORATORY CONDITIONS FOR VARIOUS TREE SPECIES

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c]		Species effect[d]	Caveat[e]	Reference
				Foliage	Yield			
65 (0.025)	6 hr	EC/SD	E. white pine	X		Threshold dose for needle damage to most sensitive clones only	--	Houston, 1974
130-390 (0.05-0.15)	6 hr	EC/SD	E. white pine	X		60% of tolerant clones developed foliar injury	Sensitive clones	Houston, 1974
130 (0.05)	16 wk Winter	EC/SD	Beech	--		Number of dead buds in spring increased at 0.10 ppm and higher; 50% more killed at 0.20 ppm.	--	Keller, 1978
130 (0.05)	10 wk	EC/SD	Norway spruce	X	X	No foliar effect; 25% less vol. growth (avg.)	2 clones only	Keller, 1980
260 (0.10)				X	X	Foliar injury; 38% less vol. growth (avg.)		
520 (0.20)				X	X	Foliar injury; 53% less vol. growth (avg.)		
470-520 (0.18-0.20)	24 hr	EC/SD	Jack pine	--		Foliar lipid synthesis inhibition (reversible); increasing dose = increasing recovery time	--	Malhotra and Kahn, 1978
660 (0.25)	2 hr	EC/SD	E. white pine	X		6.5% foliar injury	Plants maintained in sensitive condition	Berry, 1971
			Jack pine	X		4.5% foliar injury		
			Red pine	X		0.5% foliar injury		
660 (0.25)	2 hr	EC/SD	Loblolly pine	X		All equally sensitive; most sensitive period 8-10 weeks of age or older	Plants maintained in sensitive condition	Berry, 1974
			Shortleaf pine	X				
			Slash pine	X				
			Virginia pine	X				
920 (0.35)	3 hr	EC/SD	Trembling aspen	X		2% foliar injury	--	Karnosky, 1976
1180 (0.45)	6 hr	EC/SD	E. white pine	X		All tolerant clones developed foliar injury	--	Houston, 1974

TABLE 8A-3 (continued)

Conc.[a] $\mu g/m^3$ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
1180 (0.45)	9 hr/d for 8 wk	EC	Ponderosa pine	X		Severe needle tip chlorosis and necrosis	--	Evans and Miller, 1975
1310 (0.50)	2 hr	EC/SD	E. white pine Jack pine Red pine	X X X		12% foliar injury 11% foliar injury 2% foliar injury	--	Berry, 1971
1310 (0.50)	3 hr	EC/SD	Trembling aspen	X		11% foliar injury	--	Karnosky, 1976
1310 (0.50)	5 hr	GC	Austrian pine Ponderosa pine Scotch pine Balsam, Fraser fir White fir Blue, white spruce Douglas fir	X		No injury	--	Smith and Davis, 1978
1310 (0.50)	24 hr/d up to 30 d	GC	Chinese elm Gingko Norway maple Pin oak	X X X X		7 days to chlorosis 14 days to chlorosis 12 days to chlorosis 30 days to chlorosis	--	Temple, 1972
1310 (0.50)	24 hr/d 1/wk	EC/SC	Sugar maple Black oak White ash			54% lower rate of photosynthesis, no symp. 48% lower rate of photosynthesis, no symp. 20% lower rate of photosynthesis, no symp.	--	Carlson, 1979
1310 (0.50)	4 hr/d 3 wk	EC/SD	Sugar maple Black oak White ash			43% lower rate of photosynthesis, no symp. 74% lower rate of photosynthesis, no symp. 7% lower rate of photosynthesis, no symp.	--	Carlson, 1979
1700 (0.65)	3 hr	EC/SD	Trembling aspen	X		23% foliar injury	--	Karnosky, 1976

TABLE 8A-3 (continued)

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c]		Species effect[d]	Caveat[e]	Reference
				Foliage	Yield			
2620 (1.00)	4 hr	GC	Austrian pine Ponderosa pine Scotch pine Balsam, Fraser fir White fir Blue, white spruce Douglas fir	X		Less than 4% foliar injury all species	--	Smith and Davis, 1978
2620 (1.00)	8 hr	EC	American elm			Inhibition of stomatal closing	--	Noland and Kozlowski, 1979
2620 (1.00)	1 hr	EC	Scotch pine	X		No injury, primary needles; slight injury, secondary needles	--	Smith and Davis, 1977
	3 hr			X		14% maximum injury primary needles; 52% maximum injury secondary needles		
	5 hr			X		37% maximum injury primary needles; 60% maximum injury secondary needles		
5240 (2.00)	2 hr	GC	Austrian pine Ponderosa pine Scotch pine Balsam, Fraser fir White fir Blue, white spruce Douglas fir	X		No foliar injury on Douglas fir, firs, spruce Pine foliar injury threshold, necrotic tips	--	Smith and Davis, 1978
5240 (2.00)	6 hr	GC	American elm	X	X	Induced severe foliar injury; defoliation in older leaves; significant reduced expansion of new leaves; no. of emerging leaves and root dry wt. reduced	--	Constantinidou and Kozlowski, 1979a

TABLE 8A-3 (continued)

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Yield	Species effect[d]	Caveat[e]	Reference
5240 (2.00)	6 hr	GC	American elm			No significant reduction in lipid content; significant less in new leaf protein content; significant less leaf, stem root carbohydrate content	--	Constantinidou
5240 (2.00)	6 hr	GC	Chinese elm	X		100% leaf necrosis	--	Temple, 1972
5240 (2.00)	6.5 hr	EC	Gingko			Water-stressed plant increased uptake of SO_2	--	Noland and Kozlowski, 1979
5240 (2.00)	12 hr	GC	American elm			Induced stomatal closing; S content increased in plants fumigated in light	--	Temple, 1972
7860 (3.00)	6 hr	GC	Gingko Norway maple	X		50% leaf necrosis	--	Temple, 1972

[a] Table arranged by increasing SO_2 concentration as first order and exposure time as second order divisions. Doses within a single study that did not induce specifically different effects are listed along with the lowest SO_2 concentration that induced said effects.

[b] GC = growth chambers, EC = exposure chambers, EC/SD = exposure chamber, special design.

[c] X indicates study found foliar and/or yield effects.

[d] Most prominent or significant effect reported.

[e] Caveats for consideration about proper study design and interpretation.

[f] The symbol ">" means greater than; "<" means less than.

Note: 1 ppm SO_2 = 2620 ug/m³.

TABLE 8A-4. DOSE-RESPONSE INFORMATION SUMMARIZED FROM LITERATURE PERTAINING TO NATIVE PLANTS AS RELATED TO FOLIAR, YIELD, AND SPECIFIC EFFECTS BY INCREASING SO_2 DOSE

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
63 (0.024)	85 d	EC/SD	S23 Ryegrass		X	Plants at high nitrogen; SO_2 exposure alleviated S deficiency of plants not provided SO_4 and increased yield by 27%. No effect on plants grown with adequate SO_4	--	Cowling and Lockyer, 1978
					X	Plants at low nitrogen: no effect of SO_4 or SO_2		
50 (0.02)	29 d and 22 d later	EC/SD	S23 Ryegrass		X	No significant effects at 0.02 or 0.14 at first harvest (29 days);	--	Cowling and Koziol, 1978
370 (0.14)					X	At 0.14, significant reduction (16%) in specific leaf area at second harvest (22 days later). No significant effect on dry wt., tillers, dark respiration or transpiration coefficients.		
80 (0.03) 160 (0.06)	6 wk	EC/SD	Indian ricegrass		X	Insignificant increase in productivity (19%); Insignificant increase in productivity (21%); significant decrease in chlorophyll content (43%)	--	Ferenbaugh, 1978
176 (0.067)	26 wk	EC/SD	S23 ryegrass		X	Significant increase in number (88%) and dry wt. (78%) of dead leaves; significant decrease in number of tillers (41%); leaf area (51%), dry wt. of stubble (55%), and number (45%) and dry wt. (51%) of living leaves; significant decrease in yield (52%)	Preliminary study mean weekly conc. wintertime exposure	Bell and Clough, 1973
			Wild ryegrass		X	None		
210 (0.08)	13 hr/d for 28 d	EC	Foxtail grass	X	X	Foliar injury as caused by heavy metals was increased by SO_2 exposure; yield not significantly affected	No technical SO_2 monitoring information	Krause and Kaiser, 1977

TABLE 8A-4 (continued)

Conc.[a] μg/m^3 (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
290 (0.11)	4 wk	EC/SD	Cocksfoot	X	X	5% foliar necrosis; significant (30%) decrease in leaf area, dry wt. (45%), tillers, green leaves, and root/shoot ratios	Wind tunnel exposures	Ashenden, 1978
290 (0.11)	4 wk	EC/SC	Ryegrass		X	Significant (20%) decease in leaf area, dry wt. (40%), root/shoot ratio at a windspeed of 25m min^{-1} (.93mph) No effect at a windspeed of 10m min^{-1} (.37mph)	Wind tunnel exposures	Ashenden and Mansfield, 1977
290 (0.11) (weekly mean/76 μg/m^3 0.067)	103.5 hr/wk for 20 wk	EC/SD	Smooth-stalked meadowgrass		X	Significant decrease in leaf area (28%), all dry wt. fractions (44%), leaves (37%), and tillers (27%)	--	Ashenden, 1978
310 (0.12)	9 wk	EC/SD	S23 ryegrass		X	Significant decrease in dry wt. (46%) and number (34%) of living leaves, tillers (42%), leaf area (44%), dry wt. of stubble (47%); significant increase in number (46%) and dry wt. (46%) of dead leaves; significant decrease in yield	--	Bell and Clough, 1973
			Wild ryegrass		X	None		
340 (0.13)	6 wk	EC/SD	Indian ryegrass	X	X	Necrotic foliar lesions noted; insignificant decrease in productivity (6%), significant decrease in chlorophyll content (51%)	Wind tunnel exposures	Ferenbaugh, 1978
660 (0.25)						Necrotic foliar lesions noted; significant decrease in productivity (35%) and chlorophyll content (61%)		
1310 (0.50)						Plants mostly dead after 4 weeks		
2620 (1.00)						Plants dead after 4 weeks		

TABLE 8A-4 (continued)

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
390 (0.15)	6 wks		Duckweed		X	Decrease in diameter of fronds, no dry wt. effects	Ambient air + SO_2 exposure system not described	Fankhauser et al., 1976
790 (0.30)						Decrease in diameter of fronds		
1570 (0.60)						Decrease in starch content, no dry wt. effects, no irreversible damages up to 0.60 ppm SO_2		
520 (0.20)	2 hr	GC	Kentucky bluegrass	X		Visible foliar injury, high degree of variation among 17 cv's	No SO_2 monitoring information for plants previously exposed to SO_2	Murray et al., 1975
660 (0.25)	5 wk	EC/SD	Ryegrass		X	Significant decrease in yield (17%), no effect on number of tillers	Wind tunnel exposure	Horsman et al, 1978
710 (0.27)	8 wk					Significant decrease in green wt. (38%), total dry wt. (30%), no reduction in number of tillers, senescence rate doubled		
1310-28820 (0.50-11.00)	2 hr	F/CC	87 desert species	X		Most plants required more than 2.0 ppm SO_2 to produce foliar injury	Field plants watered heavily and exposed to ambient air before and after fumigation	Hill et al., 1974
1860 (0.71)	1 hr 2 hr 5 hr	EC/SD	Lily	--		Significant pollen tube elongation, inhibition at 1 and 2 hours	Pollen on agar, relationship these effects have to ambient conditions is unknown	Masaru et al., 1976

TABLE 8A-4 (continued)

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
2620 (1.0)	6 hr	EC/SD	Eucalyptus	X		40% more foliar necrosis, 32 of 131 species of Australian trees and shrubs were rated as sensitive to acute ($\geq$1 ppm) exposure to SO_2	--	O'Conner et al., 1974

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[a] Table arranged by increasing SO_2 concentration as first order and exposure time as second order divisions. Doses within a single study that did not induce specifically different effects are listed along with the lowest SO_2 concentration that induced said effect.

[b] F/CC = field, closed chambers; GC = growth chambers; EC = exposure chambers; EC/SD = exposure chamber, special design.

[c] X indicates study found foliar and/or yield effects.

[d] Most prominent or significant effect reported.

[e] Caveats for consideration about proper study design and interpretations.

[f] The symbol ">" means greater than; "<" means less than.

Note: 1 ppm SO_2 = 2620 ug/m^3.

TABLE 8A-5. EFFECTS OF MIXTURES OF SO_2 AND O_3 ON PLANTS

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Productivity	Species effect[d]	Caveat[e]	Reference
66 SO_2 + 100 O_3 (0.025 SO_2 + 0.05 O_3)	6 hr	EC/SD	E. white pine	X	X	No effect on needle elongation. Foliar injury on sensitive clones only; 10 of 10 trees with 75-100% of needles with tip necrosis. SO_2 alone caused tip necrosis on 75-100% of the needles on 1 tree. O_3 alone caused no injury.	Most O_3 meters factory-calibrated by sodium thiosulfate method	Houston, 1974
130 SO_2 + 100 O_3 (0.05 SO_2 + 0.05 O_3)	8 hr/d, 5 d/wk 5 wk	EC/SD	Radish	X	X	Plant weight reductions additive (leaf fresh and dry weight) or significantly less than additive (plant fresh wt., root fresh and dry weight).	--	Tingey et al., 1971b
130 SO_2 + 100 O_3 (0.05 SO_2 + 0.05 O_3)	8 hr/d, 5 d/wk 18 d	EC/SD	Soybean	X	X	Additive foliar injury effects; greater-than-additive root dry weight	--	Tingey et al., 1973b
130 SO_2 + 100 O_3 (0.05 SO_2 + 0.05 O_3)	8 hr/d, 5 d/wk 4 wk	EC/SD	Tobacco		X	Additive growth reductions	--	Tingey and Reinert, 1975
	8 hr/d, 5 d/wk until control plants 40-45 cm high		Alfalfa		X	Less-than-additive growth reductions	--	
160 SO_2 + 100 O_3 (0.06 SO_2 + 0.05 O_3)	7 hr/d 68 d	F/CC	Alfalfa		X	No significant alteration of plant responses (carbohydrate, protein, dry wt.) compared to effects of single pollutants	Potted plants set on soil surface; grown hydroponically	Neely et al., 1977
197-1570 SO_2 + 290 O_3 (0.075-0.60 SO_2 + 0.15 O_3)	5 or 10 d	EC/SD	White bean	X	X	Less-than-additive growth reductions and foliar injury	--	Hofstra and Ormrod, 1977
			Soybean	X		Less-than-additive foliar injury	--	

TABLE 8A-5. (continued)

Conc.[a] µg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Productivity	Species effect[d]	Caveat[e]	Reference
260 SO_2 + 200 O_3 (0.10 SO_2 + 0.10 O_3)	6 hr/d 133 d	F/CC	Soybean	X	X	SO_2 alone and in the mix did not significantly affect the yield and injury responses	--	Heagle et al., 1974
260-1310 SO_2 + 100-200 O_3 (0.10-0.50 SO_2 + 0.05-0.10 O_3)	4 hr	EC/SD	Alfalfa Broccoli Cabbage Radish Tomato Tobacco Bel-W_3	X		Greater-than-additive foliar injury at 0.10 ppm of each gas for alfalfa, broccoli, and radish. Less-than-additive effect for tomato. At 0.25 ppm, SO_2 + 0.10 ppm O_3 greater-than-additive injury noted on alfalfa, radish, and tobacco. At 0.50 ppm SO_2 and 0.05 ppm O_3, greater-than-additive injury on broccoli and tobacco and less-than-additive injury on alfalfa. At 0.50 ppm SO_2 and 0.10 ppm O_3, greater-than-additive effects on alfalfa, cabbage, radish, tobacco.	--	Tingey et al., 1973a
520 SO_2 + 100 O_3 (0.20 SO_2 + 0.05 O_3)	3 hr	EC/SD	Trembling Aspen (5 clones)	X		Greater-than-additive injury to 3 clones no injury due to SO_2 alone	--	Karnosky, 1976
920 SO_2 + 100 O_3 (0.35 SO_2 + 0.05 O_3)	3 hr	EC/SD	Trembling Aspen (5 clones)	X		Greater-than-additive injury to 4 clones	--	Karnosky, 1976
630 SO_2 + 530 O_3 (0.24 SO_2 + 0.27 O_3)	2 hr	GC	Tobacco Bel-W_3 Bel-B Consolidation 402	X		9-38% foliar injury--no injury due to either pollutant singly	--	Menser and Heggestad, 1966
730 SO_2 + 550 O_3 (0.28 SO_2 + 0.28 O_3)	4 hr	GC	Tobacco Bel-W_3 Bel-B Consolidation 402	X		23-76% foliar injury--no injury due to either pollutant singly	--	Menser and Heggestad, 1966

TABLE 8A-5. (continued)

Conc.[a] μg/m³ (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	 Productivity	Species effect[d]	Caveat[e]	Reference
660 SO_2 + 100 O_3 (0.25 SO_2 + 0.05 O_3)	4 hr	EC/SD All species exposed to both concentrations of each pollutant.	Alfalfa Onion Soybean Tobacco Bel-B Tobacco White gold	X		Only tobacco Bel-W_3 showed greater-than-additive foliar injury at 0.25 SO_2 + 0.05 O_3	Bel-W_3 tobacco very sensitive	Tingey et al., 1973a
2620 SO_2 + 200 O_3 (1.00 SO_2 + 0.10 O_3)			Tobacco Bel-W_3 Lima bean Broccoli Bromegrass Cabbage Radish Spinach Tomato			At 1.00 ppm SO_2 + 0.10 O_3 tobacco Bel-B and Bel-W_3 exhibited greater-than-additive efffects, and there were less-than-additive effects for bromegrass, cabbage, spinach, and tomato.	--	Ibid.
660 SO_2 + 490 O_3 (0.025 SO_2 + 0.25 O_3)	4 hr/d, 3 d/wk 11 wk	EC/SD	Soybean	X	X	Additive growth effects	--	Reinert and Weber, 1980
1180 SO_2 + 290 or 880 O_3 (0.45 SO_2 + 0.15 or 0.45 O_3)	4 hr	EC/SD	Radish		X	Additive growth effects	--	Tingey and Reinert, 1975
660 SO_2 + 270 O_3 (0.25 SO_2 + 0.14 O_3)	10 6hr/d 68 hr 5 d/wk 154 164	EC/SD	Scotch Pine	X		Less-than-additive effects--no effects due to O_3 alone	--	Nielsen et al. 1977
660 SO_2 + 570 O_3 (0.25 SO_2 + 0.29 O_3)	10 6hr/d 68 hr 5 d/wk 154 164	EC/SD	Scotch Pine	X		Less-than-additive effects--no effect due to O_3 alone	--	Nielsen et al. 1977

TABLE 8A-5. (continued)

Conc.[a] μg/m^3 (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c]		Species effect[d]	Caveat[e]	Reference
				Foliage	Productivity			
1310 SO_2 + 490 O_3 (0.50 SO_2 + 0.25 O_3)	4 hr/d 4 times, 6 d apart	EC/SD	Begonia (5 cv's)	X	X	Less-than-additive effects for flower weight of one cv; 0.50 ppm SO_2 alone significantly reduced flower production in the absence of foliar injury for one cv.	--	Reinert and Nelson, 1980

[a] Table arranged by increasing SO_2 concentration as first order and exposure time as second order divisions. Doses within a single study that did not induce specifically different effects are listed along with the lowest SO_2 concentration that induced said effect.

[b] F/CC = field, closed chambers; EC/SD = exposure chamber, special design; GC = growth chamber.

[c] X indicates study found foliar and/or yield effects.

[d] Most prominent or significant effect reported.

[e] Caveats for consideration about proper study design and interpretation.

Note: 1 ppm SO_2 = 2620 ug/m^3.
1 ppm O_3 = 1960 μg/m^3.

TABLE 8A-6. EFFECTS OF MIXTURES OF SO_2 AND NO_2 ON PLANTS

Conc[a] $\mu g/m^3$ ppm	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c] Foliage	Effects on[c] Yield	Species effect[d]	Caveat[e]	Reference
130-660 SO_2 + 90-470 NO_2 (0.05-0.25 SO_2 + 0.05-0.25 NO_2)	4 hr	EC/SD	Tobacco Pinto Bean Tomato Radish Oats Soybean	X X X X X X		0-2% foliar injury at 0.05 ppm SO_2+ 0.05 ppm NO_2. 1-35% foliar injury at 0.10 ppm SO_2 and 0.10 or 0.15 ppm NO_2. Injury less at 0.20-0.25 ppm SO_2 than at 0.10 ppm SO_2. Thresholds--SO_2=0.50 ppm, NO=2.00 ppm	--	Tingey et al., 1971a
390-660 SO_2 + 190-380 NO_2 (0.15-0.25 SO_2 + 0.10-0.20 NO_2)	4 hr	EC/SD	Pinto Bean Tomato Radish Oats	X X X X		No foliar injury	Recirculating air Experimental conditions approximated those of Tingey et al., 1971b, but used different cultivars of tomato, radish, and oats	Bennett et al., 1975
290 SO_2 + 210 NO_2 (0.11 SO_2 + 0.11 NO_2)	103.5 hr/wk 20 wk	EC/SD	Cocksfoot Meadow-grass		X X	Greater-than-additive decreases in no. of tillers, no. leaves, and leaf area	Wintertime exposures	Ashenden, 1979
325-2620 SO_2 + 235-1880 NO_2 (0.125-1.0 SO_2 + 0.125-1.0 NO_2)	1 hr or 3 hr	EC/SD.	Radish Swiss chard Oats Sweet pea	X X X X		Greater-than-additive foliar injury to radish at higher concentrations. Thresholds for radish were 0.50 ppm of each gas, and for the other species 0.75 ppm of each gas.	Recirculating air	Bennett et al., 1975
390-1310 SO_2 + 280-750 NO_2 (0.15-0.50 SO_2 + 0.15-0.40 NO_2)	1 hr and 2 hr	EC/SD	Alfalfa			Slight but significantly greater-than-additive depression of photosynthesis at SO_2 concentrations of 0.15 and 0.25. No greater-than-additive effects at 0.35 or 0.50 ppm SO_2.	Reversible effects	White et al., 1974
520 SO_2 + 190 or 1880 NO_2 (0.2 SO_2 + 0.1 or 1.0 NO_2)	6 d	EC/SD	Pea			Significantly greater than additive increase in peroxidase and RuDPC enzyme activity at 0.20 ppm SO_2 and and 0.10 ppm NO_2	Reversible effect	Horsman and Wellburn, 1975

TABLE 8A-6. (continued)

Conc.[a] µg/m^3 (ppm)	Exposure[a] time	Exposure[b] condition	Plant	Effects on[c]		Species effect[d]	Caveat[e]	Reference
				Foliage	Yield			
1310-28820 SO_2 + 190-940 NO_2 (0.5-11.0 SO_2 + 0.1-5.0 NO_2	2 hr	F/CC	87 desert species	X		No evidence of SO_2 + NO_2 synergism at an NO_2/SO_2 ratio of 0.28. Most species required over 5240 µg/m^3 (2.00 ppm) SO_2 to cause injury.	Plants exposed to ambient air before and after fumigation	Hill et al., 1974
2100 SO_2 + 560 NO_2 (0.8 SO_2 + 0.3 NO_2)	2 hr	EC/SD	Alfalfa	X		Apparent photosynthesis reduced at 567 mg/m^3 (315 ppm) CO_2 but increased at 1161 mg/m^3 (645 ppm) CO_2	Reversible effects	Hou et al., 1977

[a] Table arranged by increasing SO_2 conrentration as first order and exposure time as second order divisions. Doses within a single study that did not induce specifically different effects are listed along with the lowest SO_2 concentration that induced said effect.

[b] EC/SD = exposure chamber, special design; F/CC = field, closed chambers.

[c] X indicates study found foliar and/or yield effects.

[d] Most prominent or significant effect reported.

[e] Caveats for consideration about proper study design and interpretation.

Note: 1 ppm SO_2 = 2620 ug/m^3.
1 ppm NO_2 = 1885 µg/m^3.
1 ppm CO_2 = 1.8 mg/m^3.

9. EFFECTS ON VISIBILITY AND CLIMATE

9.1 INTRODUCTION

Pollutants can change the way we see the world and the climate in which we live. Because these changes may have profound consequences for the quality of life, the relationships among air quality, visibility, and climate must be understood. Although some questions about these relationships remain unanswered, many have been resolved. To provide a framework for understanding effects of pollutants on visibility, this chapter discusses factors that affect visibility, ways to measure it, historical trends, and methods to determine its value to us. Climatic effects are given similar but briefer treatment, since theoretical understanding in this area is far less advanced.

Many complex physical and chemical atmospheric processes affect our ability to see distant objects or to distinguish nearby objects clearly. For example, small particles, which are invisible individually, collectively reduce the visual range. We easily understand the visibility reduction caused by common phenomena such as rain, dust, or snow, but atmospheric constituents that we cannot perceive directly without instruments also interfere with visual range. These constituents are known as fine PM and do not include SO_2. But since SO_2 forms particulate sulfate in the atmosphere, it affects visibility indirectly.

Through both light scattering and absorption, airborne particles reduce visibility and thereby affect transportation safety and aesthetic vistas. While human observers are remarkably sensitive to contrast, they are not ideal detectors. If pollution sufficiently reduces the contrast between an object and its background, the only way a person can distinguish between them is to move closer. For the operator of a high-speed vehicle, this loss of visibility may be fatal, given the limits of human reaction time.

Scientists have made progress in evaluating the optical changes and perceptual consequences due to increased air pollution, but it has proved more difficult to evaluate the effects of pollution upon the aesthetic appeal of the environment. Economists have measured the loss of aesthetic appeal in market simulation studies with only limited success. While precise measurement of this factor remains problematic, reduced aesthetic appeal unquestionably carries significant social and economic cost. This is true for geographic areas of both good and poor visibility. More work is needed in evaluating the consequences of visibility reduction.

The effects of light scattering and absorption by PM on climate may be threefold. First, reduction of solar radiation at ground level makes less energy available for photosynthesis and commercial exploitation of solar energy. Second, reductions in solar radiation may lead to alterations in local or regional temperatures. Third, increased cloud formation may alter precipitation patterns. These effects, however, cannot yet be quantitatively related to pollutant emissions or concentrations.

General visibility conditions in the United States can be understood by examining available regional airport visibility data. The isopleths in Figures 9-1 and 9-2 (Trijonis and Shapland, 1979) show median yearly and summer visibility, a statistic that is relatively insensitive to the site-specific availability of markers as long as the farthest markers consistently reported lie beyond the median visibility. The data represent midday visual ranges for 1974-1976 from 100 suburban/nonurban locations. Visibility at 93 of the locations was determined from airport observations, usually made by the National Weather Service. Instrumental visibility measurements from seven sites in the Southwest are also included. Although some uncertainties arise from the use of airport visibility observations, there is reasonably good consistency among airport observations within regions and between airport and instrumental results in the Southwest.

The best visibility (110+ kilometers, 70+ miles) occurs in the mountainous Southwest. Visibility is also quite good (70 to 110 kilometers, 45 to 70 miles) north and south of that region, but sharp gradients occur to the east and west. Most of the area east of the Mississippi and south of the Great Lakes has a median visibility of less than 24 kilometers (15 miles).

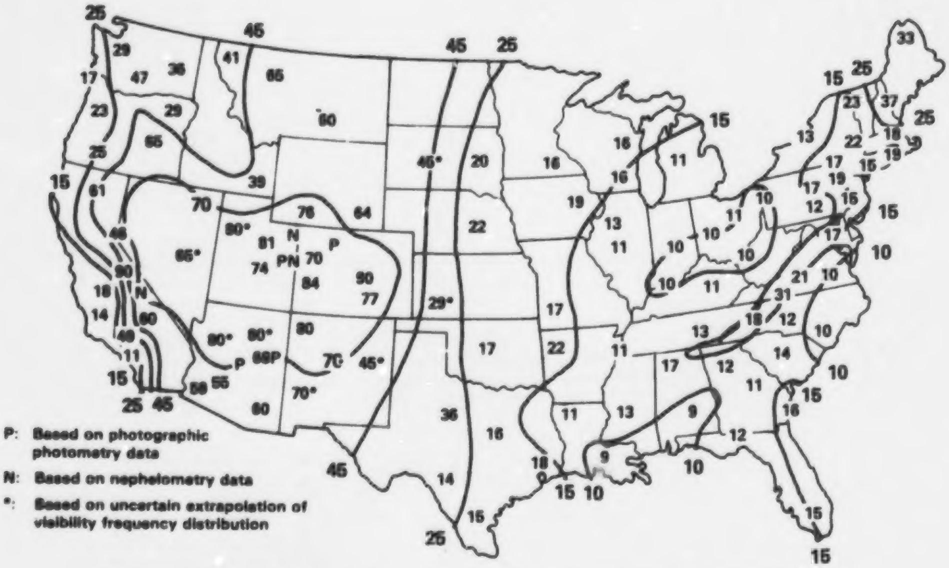
9.2 FUNDAMENTALS OF ATMOSPHERIC VISIBILITY

Figure 9-3(A) shows the simple case of a beam of light (e.g., from the sun or a searchlight) transmitted horizontally through the atmosphere. The intensity of the beam in the direction of the observer, I, decreases with distance from the source as light is absorbed or scattered out of the beam. Over a short interval, this decrease is proportional to the length of the interval and to the intensity of the beam at that point:

$$-dI = \sigma_{ext}\, I\, dx \qquad (9\text{-}1)$$

where -dI is the decrease in intensity, σ_{ext} is the extinction coefficient (the proportionality constant), I is the intensity of the beam at the beginning of the interval, and dx is the interval length. The extinction coefficient has units of inverse length. The extinction coefficient is determined by the scattering and absorption of light by particles and gases and varies with particle and gas concentration, particle size distribution and composition, and wavelength of light.

Consider an observer looking at a distant target in the daytime (Figure 9-3(B)). Just as a light beam is attenuated by the atmosphere, the light from the target reaching the observer is diminished by absorption and scattering. In addition, the observer receives extraneous light (often called air light) scattered into the line of sight by the intervening atmosphere. The net effect, as Figure 9-4 shows, is that a target darker than the horizon appears brighter than it actually is and a target brighter than the horizon appears darker than it actually is.



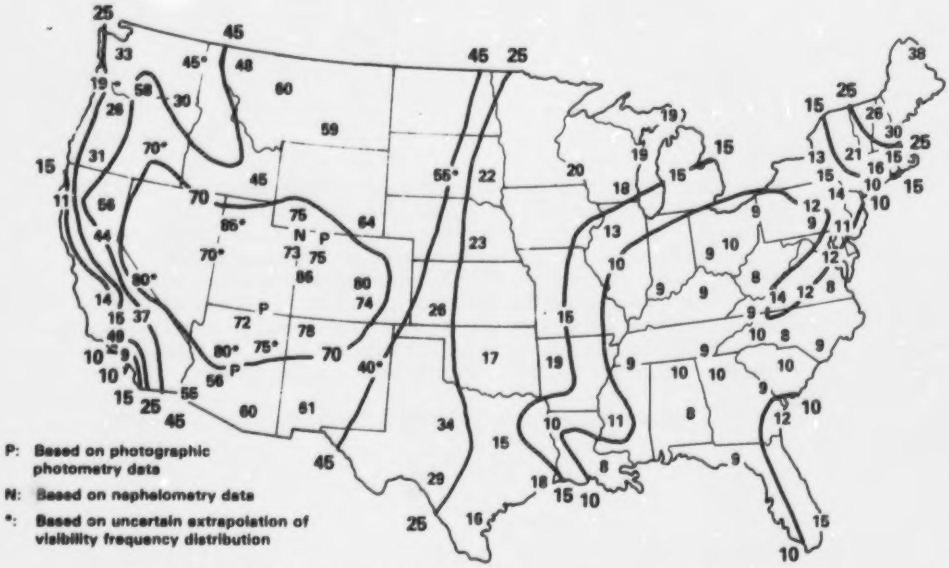
P: Based on photographic photometry data

N: Based on nephelometry data

*: Based on uncertain extrapolation of visibility frequency distribution

Figure 9-1. Map shows median yearly visual range (miles) and isopleths for suburban/nonurban areas, 1974-76.

Source: Trijonis and Shapland (1979).



P: Based on photographic photometry data

N: Based on nephelometry data

*: Based on uncertain extrapolation of visibility frequency distribution

Figure 9-2. Median summer visual range (miles) and isopleths for suburban/nonurban areas, 1974-76.

Source: Trijonis and Shapland (1979).

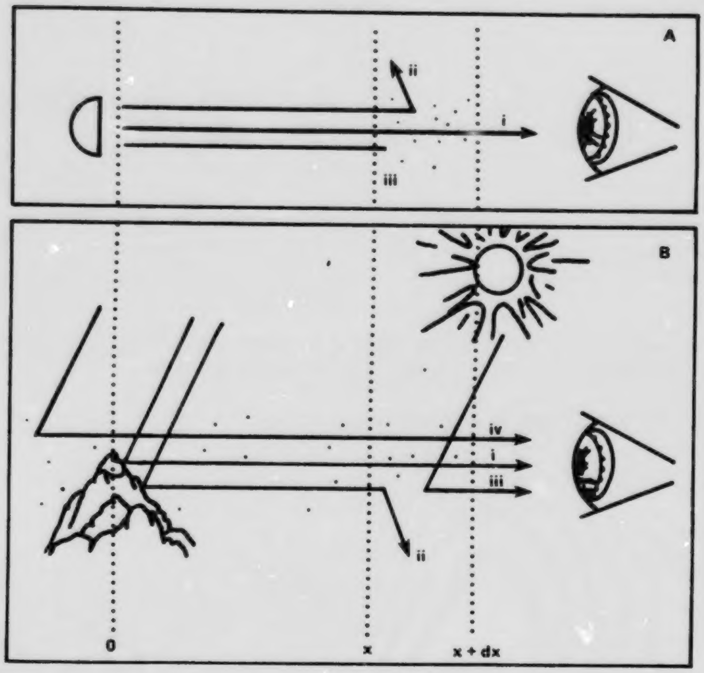


Figure 9-3. (A) A schematic representation of atmospheric extinction, illustrates (i) transmitted, (ii) scattered, and (iii) absorbed light. (B) A schematic representation of daytime visibility illustrates: (i) light from target reaching observer, (ii) light from target scattered out of observer's line of sight, (iii) air light from intervening atmosphere, and (iv) air light constituting horizon sky. (For simplicity, diffuse illumination from sky and surface is not shown). The extinction of transmitted light attenuates the "signal" from the target at the same time as the scattering of air light is increasing the background "noise."

Source: Adapted from U.S. Environmental Protection Agency (1979).

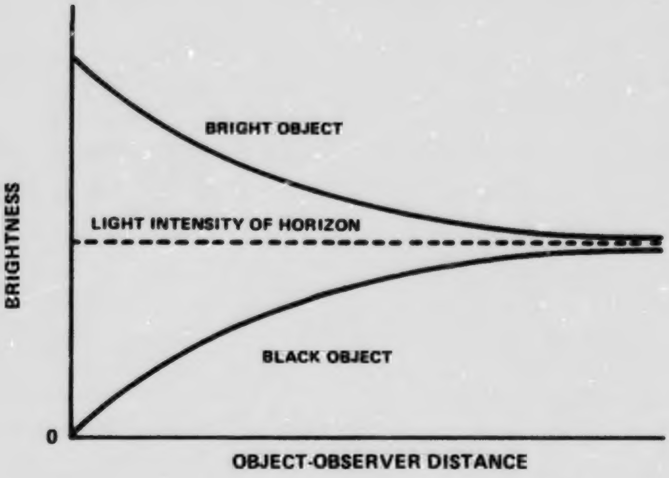


Figure 9-4. The apparent contrast between object and horizon sky decreases with increasing distance from the target. This is true for both bright and dark objects.

Source: Middleton, (1952).

At increasing distances, the apparent brightness of dark and bright targets approaches the horizon brightness. At sufficient distance, the target and horizon are so close in brightness that the target is indistinguishable; for a black target this distance is the visual range, or visibility.

The example just discussed can be restated in terms of a target's contrast, defined as:

$$C = \frac{L_{target}}{L_{background}} - 1 \qquad (9\text{-}2)$$

where L is luminance (brightness).

A target's apparent contrast begins with some intrinsic value at the target and approaches zero as distance from the target increases. When the observer's contrast threshold is reached, the target cannot be distinguished and for black targets the visual range is known.

In a uniform atmosphere, the apparent contrast between a target and the horizon sky decreases exponentially with observer-target distance x (Middleton, 1952):

$$C = C_0 \exp(-\sigma_{ext}\, x) \qquad (9\text{-}3)$$

where C_0 is the intrinsic contrast at x = 0. The maximum distance at which a given large target can be distinguished from the horizon sky is, therefore, inversely proportional to the extinction coefficient:

$$x_{max} = \frac{\log_e C_0 - \log_e \varepsilon}{\sigma_{ext}} \qquad (9\text{-}4)$$

where ε is the observer's contrast threshold.

The numerator in Equation 9-4 depends on the target's intrinsic contrast with the horizon and on the observer's contrast threshold. For a black target, $C_0 = -1$, so that $\log_e C_0 = 0$ and the numerator reduces to $-\log_e \varepsilon$. The visual range (V) of a black target in a uniform atmosphere is thus given by the Koschmieder (1924) formula:

$$V = \frac{-\log_e \varepsilon}{\sigma_{ext}} = \frac{K}{\sigma_{ext}} \qquad (9\text{-}5)$$

Although Equation 9-5 is based on use of a black target, little error is introduced into the determination of visual range by using as targets such nonblack features as dark forests and deep shadows. If a target's intrinsic brightness is as much as 30 percent that of the horizon sky, then the limiting distance given by Equation 9-4 is within 10 percent of the visual range. On the other hand, the intrinsic brightness of artificial lights or sunlit objects can approach that of the horizon sky, in which case their use as targets can lead to large underestimates of visual range. It must also be noted that nonblack targets can lead to serious underestimates of visual range if a target at a distance significantly less than the visual range is used and visual range is calculated or estimated from the apparent contrast of the target. For example, for an intrinsic target brightness of 30 percent of the horizon brightness, an error of 18 percent results from using a target at half the true visual range; an error of 91 percent results if the target is at one-tenth the true visual range.

The observer's contrast threshold is not a constant (Figure 9-5), but varies with the individual observer (Middleton, 1952); apparent target size (Blackwell, 1946); spatial frequency (Campbell and Maffei, 1974); brightness adaptation of the eye (Blackwell, 1946); whether the target is stationary or moving; viewing time; background uniformity and complexity; and target shape (Taylor, 1964). Neither is the contrast threshold a sharp boundary between detection and nondetection; rather, it is usually defined as the value of contrast that results in a 50-percent detection probability. Applications that require high detection probabilities (e.g., aircraft operation) often double the value (Taylor, 1964). The value of 0.02 has been widely used for many years, beginning with Koschmieder (Middleton, 1952). It is clear from the above considerations, however, that no fixed value of contrast threshold is generally applicable. Fortunately, as can be seen in Equation 9-4, uncertainties in ε and C_0 are reduced by the logarithmic function; thus, the relationship of visual range to σ_{ext} is not directly proportional to variations of ε and C_0.

Surprisingly, a target's chromaticity (hue and saturation) has only a marginal effect on changing an observer's contrast threshold and thus the visual range (Middleton, 1950). This is because the apparent color of a target near the visual range is gray, due to scattering out of the beam of colored target light and scattering into the beam of essentially white air light. Furthermore, the detection threshold of chromaticity difference increases greatly for targets of apparent angular diameter less than half a degree, angles that most brightly painted objects (e.g., radio towers) subtend. Thus, at the visual range colored targets appear about the same as gray targets of the same luminance. This is not to say that target chromaticity is unimportant in judging visibility degradation by observing targets closer than the visual range. To the contrary, washed-out colors and details of relatively nearby objects are probably the most noticeable facet of visibility degradation to observers of scenic vistas (U.S. Environmental Protection Agency, 1979).

The most direct method of determining the proportionality constant, K, involves instrumentally measuring the apparent contrast of a target that is marginally perceptible to an unaided observer. This method has been used both in laboratory determinations (Blackwell, 1946) and in atmospheric studies (Middleton, 1952).

Another way to determine K is to measure the visual range and its associated σ_{ext} and

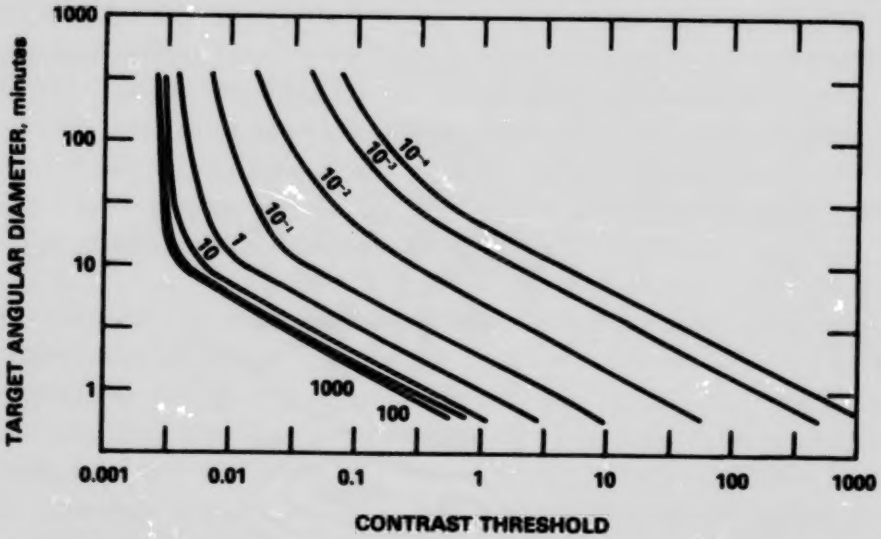


Figure 9-5. Mean contrast threshold of the human eye for 50% detection probability as a function of target angular diameter and adaptation brightness (candles/m²) for targets brighter than their background. Daytime adaptation brightness is usually in the range of 100—10,000 candles/m².

Source: Middleton (1952).

take their product. It is imperative that: (1) the intrinsic contrast of each target be known and corrected for; (2) targets be sufficiently numerous and well-spaced so as to give reasonable resolution of the visual range; (3) total extinction be measured; (4) the extinction measurement be representative of that along the sight paths to the targets; and (5) aerosol sampling for the extinction measurement be conducted carefully and detailed measurement conditions be recorded. Observations at small solar scattering angles should probably be excluded unless the method for determining σ_{ext} responds to the increased solar scattering as well.

A third approach for obtaining K, multiplying visual range by the fine mass concentration (FMC) and an assumed σ_{ext}/FMC ratio, is much less desirable due to the additional uncertainty of σ_{ext}/FMC ratio discussed in Section 9.2.3.3.

Several studies have shown substantial deviation from the historical K value of 3.9, which results from assuming large perfectly black targets and a contrast threshold of 0.02. Studies by Horvath and Noll (1969), Samuels et al. (1973), and Griffing (1980) can be criticized because they assumed absorption by particles was negligible. It is now known that particle absorption is rarely negligible, thus their estimates of K may be low by 10 to 50 percent. Griffing's work can also be criticized because of the poor quality of markers available at the airport he studied. The only study that reasonably adheres to the recommendations of the earlier paragraph is that of Ferman et al. (1981), which reports a value of 3.5 for well-mixed periods. This value may be corrected to 3.7 to account for the imperfect blackness of the tree-covered mountain targets, assuming the trees to be 20 percent as bright as the horizon sky (Middleton, 1952).

It is to be emphasized that no fixed value of K is generally applicable. As Middleton (1952) reviewed, contrast thresholds from 0.0077 to 0.06 were measured over the years, corresponding to K ranging from 4.9 to 2.8, respectively. Given the observed range of values, it seems reasonable to retain the historical choice of K = 3.9 when a value must be chosen for illustration. Users of "visual range" data derived from non-black targets must remember that they do not have true visual ranges until they correct the data for non-zero intrinsic target brightness.

The Koschmieder formula's neglect of both pollution gradients and the earth's curvature and topography limit its applicability near sources of primary PM and in very clean air. The formula also assumes equal illumination of all parts of the atmosphere in the horizontal plane, which limits its usefulness under partly cloudy skies. Where visibility is restricted by diffuse haze, however, Koschmieder's formula performs well. Comparisons of daytime visual range, measured by a human observer, and extinction from scattering, measured instrumentally at a single point, show visual range to correlate with the reciprocal of extinction, as illustrated in Figure 9-6 (Horvath and Noll, 1969; Samuels et al., 1973). The correlation co-

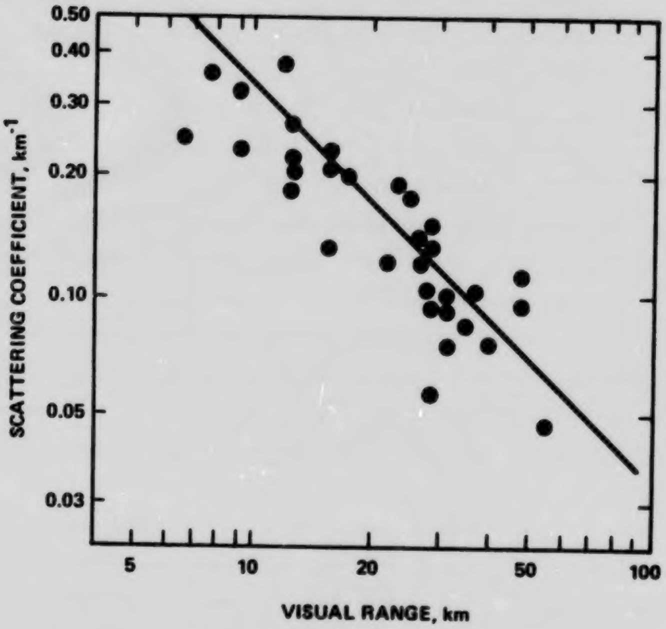


Figure 9-6. Inverse proportionality between visual range (V) and the scattering coefficient)σ_{sp}) as measured at the point of observation. The straight line is derived from the Koschmieder formula for visual range, assuming $V = 3.9/\sigma_{sp}$ and nonabsorbing media ($\sigma_{ext} = \sigma_{sp}$). The correlation coefficient for V and σ_{sp} is −0.89.

Source: Horvath and Noll (1969).

efficients are commonly in the neighborhood of 0.9, which is quite good considering that the point measurement of extinction is being extrapolated along a sight path several tens of kilometers long. In summary, visibility is inversely proportional to the atmospheric extinction coefficient.

9.2.1 Physics of Light Extinction

Since light is electromagnetic radiation, it is altered by the interaction of its electric and magnetic fields with all matter through which and near which it passes. It may be altered by scattering (redirection) or by absorption. Two areas of theoretical investigation have led to a thorough understanding of the physics of light extinction in the atmosphere. The first, derived by Lord Rayleigh in the late 1800's, pertains to the scattering of light by the gaseous molecules comprising the atmosphere (Middleton, 1952; Kerker, 1969). Rayleigh scattering is directly proportional to the molecular number density and thus decreases as elevation increases. It is inversely proportional to the fourth power of wavelength (color); thus, blue light is scattered by air molecules about five times more than is red light. Rayleigh scattering is of limited interest in visibility work because it represents the cleanest possible condition of the atmosphere. Almost all cases of visibility impairment are caused by the presence of particles, the sole exception being discoloration caused by the pollutant gas NO_2.

The second theory pertains to the extinction of light by homogeneous spheres of arbitrary size and was advanced by several workers from 1890 to 1910; the name of Mie (1908) is most commonly associated with this theory (Kerker, 1969; Middleton, 1952). This approach, based on electromagnetic theory, is necessary because resonances occur in scattering by particles of diameter within about an order of magnitude of the wavelength of the radiation. For visible light, the size range requiring use of Mie theory extends from 0.05 μm to 10 μm (and larger if the particles are spherical), the sizes at which most long-lived atmospheric particles accumulate. Mie theory allows computation of the scattering in any or all directions and the absorption of light by a single spherical particle. The wavelength of light, particle size, and complex refractive index must be specified. For absorbing particles, the imaginary part of the refractive index is nonzero. For a monodisperse aerosol of number concentration N, the extinction coefficient becomes:

$$\sigma_{ext} = N\, Q_{ext} \pi\, r^2 \qquad (9\text{-}6)$$

where r is the particle radius and Q_{ext} is the extinction efficiency factor calculated from Mie theory. Examples of the behavior of Q_{ext} are shown in Figures 9-7 and 9-8. Note that after its large rise at a few tenths of a micrometer, Q_{ext} approaches 2 and the extinction by a single particle becomes proportional to r^2.

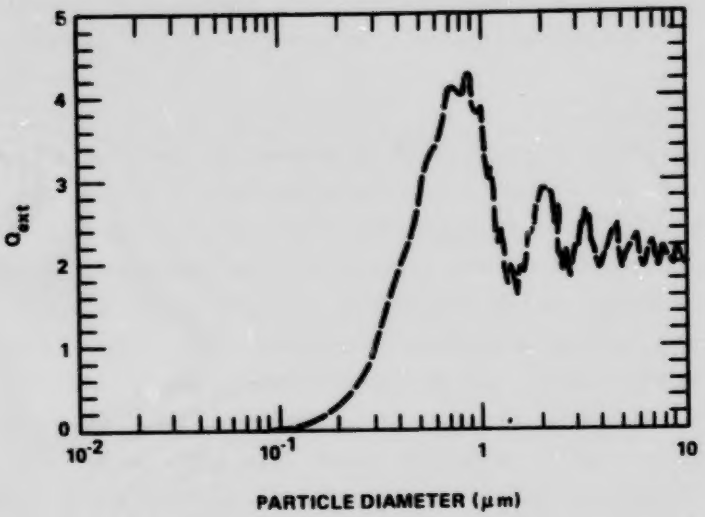


Figure 9-7. Extinction efficiency factor (Q_{ext}) of a single spherical particle as a function of diameter for a non-absorbing particle of refractive index (1.5—0.0i) and wavelength 0.55 μm.

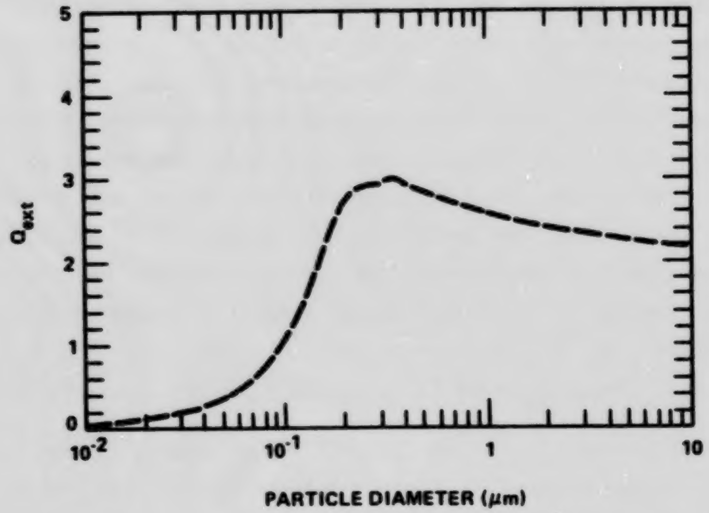


Figure 9-8. Extinction efficiency factor (Q_{ext}) of a single spherical particle as a function of diameter for an absorbing particle of refractive index (2.0—i) and wavelength 0.55 μm.

For polydisperse aerosols, integration over the range of sizes is necessary:

$$\sigma_{ext} = \pi \int_{r_1}^{r_2} n(r)\, Q_{ext}(r)\, r^2\, dr \qquad (9\text{-}7)$$

If the particles are of differing composition (and thus differing complex refractive index), the number size distribution of each species must be known. Extinction coefficients for the various species are then summed. The precise calculation of extinction by spherical particles allowed by Mie theory demands detailed knowledge of the composition and size distribution of the particles.

Because of the wavelength dependence of extinction, particles may cause discoloration of scenes. The type of discoloration (e.g., a red hue or a blue hue) depends on the scattering angle (i.e., the angle between lines of illumination and observation), the particle number distribution, the particle refractive index distribution (i.e. composition as a function of size), and the background sky and objects. The wavelength (λ) dependence of scattering is given by $\lambda^{-\alpha}$, where α ranges from 4 for extremely small particles (i.e., the Rayleigh limit) to zero for very large particles (e.g., 10-μm diameter cloud droplets); for intermediate sizes where most long-lived atmospheric aerosol particles accumulate, α ranges from 0.5 to 2.5, with 1.5 being typical (Charlson et al., 1972). The wavelength dependence of absorption is usually found to approximate λ^{-1} for atmospheric aerosol, indicative of fine particles having a small dispersion in the complex refractive index over the visible wavelengths (Weiss et al., 1978). For particles larger than a few micrometers, however, absorption becomes wavelength independent.

The intensity of scattering varies with the scattering angle (i.e., the angle between initial and scattered directions). Particles of diameter much larger than the wavelength scatter almost all light into the forward direction. At particle diameters comparable to visible wavelengths (diameters at which particles of the most interest accumulate), a small but significant fraction of light is scattered into wider angles. The exact values can be calculated from Mie theory for a given size distribution, refractive index, and wavelength. The practical significance of this angular dependence of scattering is obvious when viewing objects toward the sun: contrast, colors, and details can appear extremely degraded compared to objects in the opposite direction; plumes that appeared colored from one angle may appear gray or white against the sun, their color washed out by the addition of large amounts of essentially white light.

Mie theory is strictly applicable only to spherical particles. Fortunately, a majority of scattering particles in the optically important size range of 0.1- to 2-μm diameter are spherical (many being droplets), or nearly spherical (Allen et al., 1979; Pueschel and Wollman, 1978; Pueschel and Allee, 1980). Electromagnetic solutions have been achieved for several nonspherical shapes, such as spheroids (Latimer, 1980); disks (Weil and Chu, 1980); cylinders; and for the important case of layered or coated spheres (Kerker, 1969; Schuerman, 1980; Pinnick et al., 1976; Fowler and Sung, 1979; Mugnai and Wiscombe, 1980). These exact solutions can be applied if circumstances warrant. Absorbing particles are usually distinctly nonspherical. Many are chain-aggregates such as flame soot. Janzen (1980) empirically showed that Mie theory predicts measured absorption quite well for carbon black particles by assuming the aggregates to be spheres of equal volume.

Multiple scattering of photons (i.e., more than one scattering event en route from object to observer) is negligible for our application of Mie theory because of the relatively small particle number concentrations and extinction cross sections involved in even the worst cases of anthropogenic visibility impairment.

Light extinction by particles is determined by the number, size, shape, and type of particles in a given volume and does not vary with fluctuations or differences of ambient temperature or pressure It is recommended that mass concentration data collected at non-standard temperature or pressure not be corrected to standard or normal conditions (as is sometimes routinely done with air pollution data) for comparison with extinction values. Many techniques for measuring size distribution (e.g., impaction, laser Doppler velocimetry) measure aerodynamic diameter, not geometric diameter that is needed for Mie calculations. These diameters are related by the square root of density for a given spherical particle, but the usually unknown variation of density may cause errors of 20 percent or more in the geometric size needed for Mie calculations.

Despite the relatively advanced state of knowledge of the physics of light extinction, there are two remaining difficulties in visibility modeling:

1. How to model the transport and transformation of pollutants and precursors from sources so that the concentration, size, and type of particles along a given sight path can be predicted,
2. How to model the perception and evaluation of a scene by an observer.

The former problem is of considerable interest for reasons other than visibility and is the subject of considerable research (see chapter 6). The latter problem encompasses the disciplines of physiology, psychology, and economics, and requires an empirical approach.

9.2.2 Measurement Methods

The term "visibility" is often used to mean visual air quality, which takes into account not only how far one can see, but also how well one can see nearby objects (i.e., their apparent contrast or discoloration). These multiple meanings have confounded the choice of a particular method for measuring visual air quality.

Human observer methods have been widely used, and their qualitative judgments are the ultimate reference for any evaluation of visual air quality. Nonobserver or instrumental methods of visibility measurement, however, are very important in understanding visibility impairment for a variety of reasons:

1. They may measure separate components of extinction (e.g., only scattering or only absorption);
2. They may allow elimination of undesirable effects (e.g., increased contrast loss because of looking toward the sun);
3. They may allow modification of an air sample to check for characteristic responses of particular pollutants (e.g., humidifying a nephelometer sample to check for deliquescence);
4. They may allow statistical analysis with particle composition data collected at a site.

In many cases it is not the method that is deficient, but its application to the question at hand. If reasonable correspondence can be demonstrated between certain instrumental methods and human perception, it will probably be desirable to choose instruments for routine visibility data collection, for we would then be inferring an average perception based upon the accepted statistical link rather than the specific perception of a particular observer. In addition, the response of observer methods, photography, and telephotometry to nonpollutant conditions such as precipitation, snow on the ground, clouds, and the solar angle may limit their use.

In general, correspondence between human observation and instrumental methods is good. For specific comparisons, poor correlation is usually explainable as being due to humidity effects or poor instrument techniques or application. Investigators have usually found contrast methods (e.g., telephotometry and photography) to agree best with human observation (Allard and Tombach, 1980; Cwalinski et al., 1975). This finding is not surprising, since the eye-brain system itself relies on contrast. The consistently superior correlation of contrast methods with human observation is probably because of their common response to nonpollutant conditions such as clouds and the solar angle, as noted above. These studies and another (Horvath and Noll, 1969) have also shown that at RH below 60 to 70 percent, nephelometry compares well with human observation. The fact that these investigators had to exclude high humidity cases to achieve good agreement indicates that nephelometry is not inherently at fault; the problem arises from failure to prevent heating or cooling of the aerosol during aspiration.

Two other studies, one experimental (Hoffmann and Kuehnemann, 1979) and one theoretical (Harrison, 1979), have shown that the nephelometer provides reasonable estimates of visual range, at least for turbid and polluted atmospheres. Harrison (1979) calculated that in clear atmospheres, nephelometers may underestimate visual range by as much as 35 percent. There have been too few careful studies comparing transmissometry results and human observations to establish a credible relationship.

The extinction coefficient introduced in Section 9.2 is a summation of contributions from scattering and absorption by gases and by particles:

$$\sigma_{ext} = \sigma_{sg} + \sigma_{ag} + \sigma_{sp} + \sigma_{ap} \qquad (9\text{-}8)$$

where σ_{sg} is scattering by gas molecules (Rayleigh scatter), σ_{ag} is absorption by gases, σ_{sp} is scattering by particles, and σ_{ap} is absorption by particles. A number of methods that measure or allow estimates of the various terms of this equation are in use. A brief discussion of the more common methods follows.

9.2.2.1 Human Observer (Total Extinction)--Human observers have been most often used to determine the visual range (Middleton, 1952). In standard practice, a set of targets at known distances is selected and an observer then records whether or not he or she can see each target. The "prevailing visibility" is then defined as the greatest distance attained or surpassed around at least half of the horizon circle, but not necessarily in continuous sectors (National Weather Service, 1979). Visual range is affected not only by the optical properties of the atmosphere, but also by target characteristics, illumination conditions, and the observer (Duntley, 1964). Observer methods are labor intensive, subjective, and often must use ill-placed or nonideal targets. Visual range data may be converted to σ_{ext} if one accepts certain assumptions (Middleton, 1952) by Koschmieder's formula:

$$\sigma_{ext} = \frac{3.9}{V} \qquad (9\text{-}9)$$

The human observer also can make qualitative statements about overall visual air quality, unusual coloration, and the presence of plumes. Different days or studies are difficult to compare because even minute changes of scenes from day to day may affect human perception. This shortcoming can be partially resolved by photography.

9.2.2.2 Photography (Total Extinction)--Photographs can be used to document scenes for later qualitative analysis by humans or for later analysis of a target's apparent contrast by film densitometry (Steffans, 1949; Veress, 1972). Photographs provide more accurate long-term retention of a scene than does the human mind and enable large numbers of people to evaluate a given scene for perception studies. Photography, however, may introduce significant errors due to varying film characteristics, use of filters, exposure and processing, aging, storage conditions, and reproducibility of the image. Whenever photographs are to be compared, it is imperative to ensure that all photographic apparatus is clean; identical in the use of filters and film; and that exposures and processing are as uniform as possible. If the reproduced image of a scene is to portray accurately what a human eye sees, it is necessary that the overall response of the photographic process be photopic (i.e., match the wavelength response of the human eye); otherwise, the rendition may not be true, and both densitometry and qualitative applications may be seriously affected.

9.2.2.3 Telephotometry (Total Extinction)--A telephotometer is a telescope that can measure the apparent brightness of a faraway object (Middleton, 1952; Ellestad and Speer, 1981). By measuring the brightness of an object at distance x and the horizon sky around it, the object's apparent contrast can be computed. This number may be used directly or, if Koschmieder's restrictions are assumed (Middleton, 1952), may be converted to σ_{ext} by the equation:

$$\sigma_{ext} = \frac{-1}{x} \log_e \frac{C}{C_0} \qquad (9\text{-}10)$$

Telephotometry is attractive for several reasons. It is a path measurement; thus, atmospheric nonuniformities are averaged. The instrument's absolute calibration is unimportant; only its linearity and short-term stability matter. It requires no sample aspiration, and therefore avoids large particle losses and sample heating or cooling. Finally, it is perhaps the closest instrumental approximation to human observation. The method has limitations when the target's intrinsic contrast is unknown or assumed (a very common situation), when measuring dark objects at close range (due to internal stray light errors), and when clouds cause uneven illumination. The cloud problem makes telephotometry difficult to automate and thus the method remains labor intensive.

9.2.2.4 Long-path Extinction (Total Extinction)--The most direct way to measure σ_{ext} is to measure the decrease in intensity of a light beam over a known distance x,

$$\sigma_{ext} = \frac{-1}{x} \log_e \frac{I}{I_o} \qquad (9\text{-}11)$$

where I and I_0 are the final and initial intensities, respectively. The method is appealing in that no assumptions are involved, it measures average extinction over the path, and it requires no sample aspiration. Unfortunately, even for values of σ_{ext} about 0.2 km^{-1} (typical in the eastern United States), the decrease over short paths (1 meter) is small (0.02 percent) and cannot be measured accurately. An alternative is to increase the path length, but source intensity fluctuation, mirror reflectivity changes for single-ended systems, detector sensitivity drift, alinement, thermally induced scintillation, and the large background light of daytime again make the measurement difficult.

Hall and Riley (1976) have measured extinction by observing an uncollimated source at two ranges. Any decrease in intensity with increasing range in excess of the inverse-square decrease is due to extinction. This method avoids any need for absolute calibration, since only the ratio of intensities at two ranges need be measured. This method as used by Hall and Riley is labor intensive, but has been demonstrated in clean and urban environments during night operation.

9.2.2.5 Nephelometer (Scattering)--The integrating nephelometer measures only the scattering coefficient of an aerosol (Beuttell and Brewer, 1949; Crosby and Koerber, 1963; Ruppersberg, 1964; and Charlson et al., 1967). By simple adjustment, Rayleigh scatter, σ_{sg}, can be excluded or included. The instrument consists of an enclosed volume painted black, a sensitive light detector looking through the volume, and a light source at one side of the volume. The only light reaching the detector is that scattered by gas molecules and particles within the volume. The nephelometer is sensitive, easily calibrated and automated, enables one to modify the sample if desired, and provides a point measurement for correlation analysis with point measurements of mass concentration and chemical composition. Errors of application of the nephelometer include siting where the aerosol is nonuniform (e.g., near aerosol sources or in mountainous terrain) or siting where significant absorption may occur (e.g. urban sites), unless a concurrent measurement of absorption is made. The operator must be careful to avoid inadvertent heating or cooling of the sample and resulting modification of the aerosol size distribution. The nephelometer has two sources of inherent errors: (1) angular truncation (Ensor and Waggoner, 1970; Rabinoff and Herman, 1973), which results in underestimation of scattering, especially when large particles are present; and (2) sample aspiration (Heintzenberg and Quenzel, 1973), which results in the loss of large particles through impaction on the ductwork. These inherent errors may result in depressed correlations between scattering and total mass concentration when significant large particle concentrations occur. Despite these limitations, nephelometry remains one of the most widely used visibility measurement methods. Most of the data relating cause and effect (i.e., particle concentration or composition and optical effect) have been acquired with the nephelometer. Agreement between the nephelometer and the long-path transmission method has been demonstrated in several cases (e.g., Waggoner and Charlson, 1976; Weiss et al., 1979).

9.2.2.6 Light Absorption Coefficient--Were it not for the fact that elemental carbon (soot, graphitic C, free C) is a prominent species in cities and industrial regions, σ_{ap} would be inconsequential. Even a few percent of the submicrometer mass as soot, however, produces a significant effect on σ_{ap} or σ_{ext}. The methods that have so far been used to evaluate σ_{ap} include:

1. Determining the difference between σ_{ext} and σ_{sp} by using long-path transmissometry and nephelometry (Weiss et al., 1979);
2. Determining change of transmission of Nuclepore® filters with scattered light collected by an integrating plate of opal glass (Lin et al, 1973; Weiss et al., 1979);
3. Determining change of transmission of Millipore® filters (Rosen et al., 1980);

4. Determining the reflectivity of a white powder with aerosol mixed into it, called the Kubelka-Monk method (Lindberg and Laude, 1974);

5. Determining absorption of light by a sample of particles inside a white sphere (integrating sphere) (Elterman, 1970);

6. Estimating an imaginary refractive index from regular scattering or polarization and size distribution (Eiden, 1971; Grams et al., 1974);

7. Measuring the amount of graphitic C and its size distribution and then calculating σ_{ap},

8. Detecting the acoustical pulse produced when energy is absorbed by particles as light and is transformed to heat (spectrophone) (Truex and Anderson, 1979).

As yet, no single method is widely accepted, although filter methods (e.g., 2 and 3, above) are simple, inexpensive, and show promise. More work is needed to learn how the various filter methods relate to each other and, more importantly, how accurately each measures the in situ σ_{ap}. The Nuclepore® method has the distinct theoretical advantage of collecting the particles in a monolayer and on an optically simple filter. This method may overestimate σ_{ap} somewhat, however, due to backscatter losses and orientation of common chain-aggregates normal to the light beam, which maximizes their absorption cross section. Filter methods employing depth filters produce greater overestimates of σ_{ap}, probably due to multiple scattering and the resulting multiple chances for each absorption site to attenuate the light beam. For example, Sadler et al. (1981) report that σ_{ap} from quartz fiber and membrane filters averages about a factor of 2.8 higher than values from Nuclepore® filters.

Recent work on the BS method indicates that, as many researchers have speculated, this method gives a rather poor indication of total or fine mass concentration and instead responds principally to absorbing particles (Edwards, 1980; Pashel and Egner, 1981). (For further discussion see Chapter 3.) Edwards' work further demonstrates empirical conversions from BS and coefficient of haze (CoH) data to σ_{ap}. If substantiated, these conversions will allow trend analysis of σ_{ap}, because BS or CoH data have been routinely monitored at many sites for several previous decades.

9.2.3 Role of Particulate Matter in Visibility Impairment

As noted earlier, the extinction coefficient comprises contributions from gas and particle scattering and absorption:

$$\sigma_{ext} = \sigma_{sg} + \sigma_{ag} + \sigma_{sp} + \sigma_{ap} \qquad (9\text{-}12)$$

This section discusses the relative magnitudes of these contributions.

9.2.3.1 Rayleigh Scattering--A particle-free atmosphere at sea level has an extinction coefficient of about 0.012 km^{-1} for green light (wavelength 0.55 μm) (Penndorf, 1957), limiting visual range to about 325 kilometers. The coefficient σ_{sg} decreases with altitude.

In some areas of the Western United States, the extinction of the atmosphere is at times essentially that of a particle-free atmosphere (Charlson et al., 1978a; Porch et al., 1970). Rayleigh scattering thus amounts to a simply definable and measurable background level of extinction with which other extinction components (such as those caused by manmade pollutants or by natural sources of particles) can be compared. At 40-kilometer visual range, a better-than-average value for the Eastern United States, Rayleigh scattering contributes only about one-eighth of the total extinction (Trijonis and Shapland, 1979). Rayleigh scattering decreases with the fourth power of wavelength, and contributes a strongly wavelength-dependent component to extinction. When Rayleigh scattering dominates, dark objects viewed at distances over several kilometers appear behind a blue haze of scattered light, and bright objects on the horizon (such as snow, clouds, or the sun) appear reddened at distances greater than about 30 kilometers. Scattering by gaseous pollutant molecules is negligible because of their low concentrations compared to N_2 and O_2; thus, variations in pollutant gas concentrations have no effect on Rayleigh scattering.

9.2.3.2 Nitrogen Dioxide Absorption--Of all common gaseous air pollutants, only NO_2 has a significant absorption band in the visible spectrum. Nitrogen dioxide strongly absorbs blue light and can color plumes or urban atmospheres red, brown, or yellow if significant concentrations and path lengths are involved. The effects of NO_2 on visibility are discussed more fully in the NO_x criteria document and by Hodkinson (1966), White and Patterson (1981), and Charlson et al. (1972). Its contribution to total extinction is in general minor. For example, in Denver, Colorado during November and December 1978, NO_2 levels contributed an average of 6 percent of total light extinction (Groblicki et al., 1981).

9.2.3.3 Particle Scattering--In general, scattering by particles accounts for 50 to 95 percent of extinction, depending on location, with urban sites in the 50- to 80-percent range and nonurban sites in the 80- to 95-percent range. The measurements of Waggoner et al. (1981), made with an integrating nephelometer and the integrating plate method, show that in urban-industrial areas particle scattering accounts for 50 to 65 percent of extinction, in urban residential areas 70 to 85 percent, and in remote areas 90 to 95 percent. A comparison reported by Weiss et al. (1979), which employed a nephelometer and a long-path extinction device, found that scattering accounts for 55 to 65 percent of extinction in urban Phoenix, Arizona, and about 95 percent on a plateau outside Flagstaff, Arizona. Wolff et al. (1980), using a nephelometer and absorption values inferred from elemental carbon loadings, produced data that show that scattering accounted for 60 to 85 percent of extinction at a variety of sites.

Fine particles (i.e., those particles of diameter less than 1 to 3 μm) usually dominate light scattering. Coarse particles are occasionally important, particularly near roadways and some industrial sites and during natural occurrences of fog and windblown dust. Charlson et al. (1978a) used Mie theory to calculate the light-scattering and absorption efficiency per unit volume concentration of particles for a typical aerosol containing some light-absorbing soot (Figure 9-9). As illustrated in the figure, particles of 0.1 to 2 μm diameter are the most efficient light scatterers. The remarkably high scattering efficiencies of these particles are illustrated by the following examples: a given mass of aerosol of 0.5-μm diameter scatters about a billion times more light than the same mass of air; a 1-mm-thick sheet of transparent material, if dispersed as 0.5-μm particles, would be sufficient to scatter 99 percent of the incident light, that is, to obscure completely vision across such an aerosol cloud.

A more revealing explanation of the usual dominance of scattering by fine particles is possible: particles smaller than 0.1 μm, though sometimes present in high numbers, are individually very inefficient at scattering light and thus contribute very little to visibility loss; particles larger than about 1 to 3 μm, though individually efficient at scattering light, usually exist in relatively small numbers and contribute only a small fraction of visibility loss.

Atmospheric particles are made up of a number of chemical compounds (see Chapter 2). All these compounds exhibit a peak scattering efficiency in the same diameter range (0.1 to 1.0 μm) calculated to be optically important in Figure 9-9. Because of differences in refractive index, however, the values of the peak efficiency and the exact particle size at which it occurs vary considerably among the compounds (Figure 9-10; Faxvog, 1975).

In Figure 9-10, note the high extinction efficiencies of C and water. As discussed in Section 9.2.4, these compounds are often significant fine mass components and are therefore often responsible for significant amounts of extinction. Figure 9-10 should not, however, be taken to present invariable, precise extinction efficiencies of the various species. It was produced with best estimates of the refractive indexes and for monodisperse particles. In reality, the species do not exist as monodispersions or in equal concentrations, and therefore their relative roles in causing extinction may vary considerably.

Measured particle size distributions can be used in conjunction with Mie theory calculations to determine the contribution of different size classes to extinction. The results of this kind of calculation are shown in Figure 9-11. The peak in scattering per unit volume concentration is at 0.3 μm, so that the fine particles dominate extinction in most cases.

Because the peak and shape of the bimodal particle mass (or volume) distribution curve can vary, the light-scattering characteristics of a given particle mass might also be expected to vary. As noted by Charlson et al. (1978a), however, for the observed range of atmospheric particle distributions, the calculated scattering coefficient per unit mass concentration is relatively uniform. Latimer et al. (1978) have determined the scattering per unit volume concentration for aerosol size distributions having several geometric standard deviations, includ-

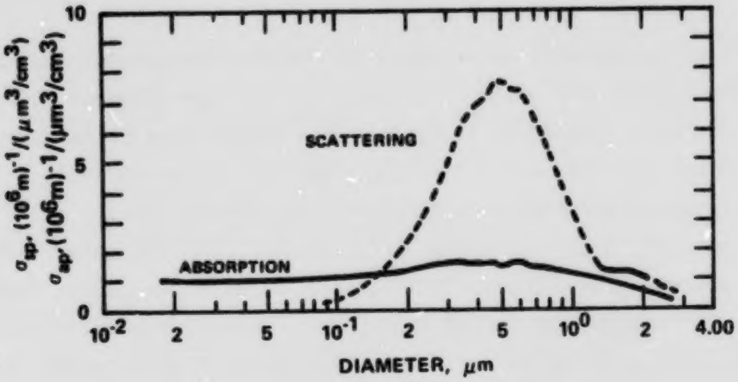


Figure 9-9. For a light-scattering and absorbing particle, the scattering per volume concentration has a strong peak at particle diameter of 0.5 μm (m = 1.5–0.05i; wavelength = 0.55 μm). However, the absorption per aerosol volume is only weakly dependent on particle size. Thus the light extinction by particles with diameter less than 0.1 μm is primarily due to absorption. Scattering for such particles is very low. A black plume of soot from an oil burner is a practical example.

Source: Charlson et al. (1978a).

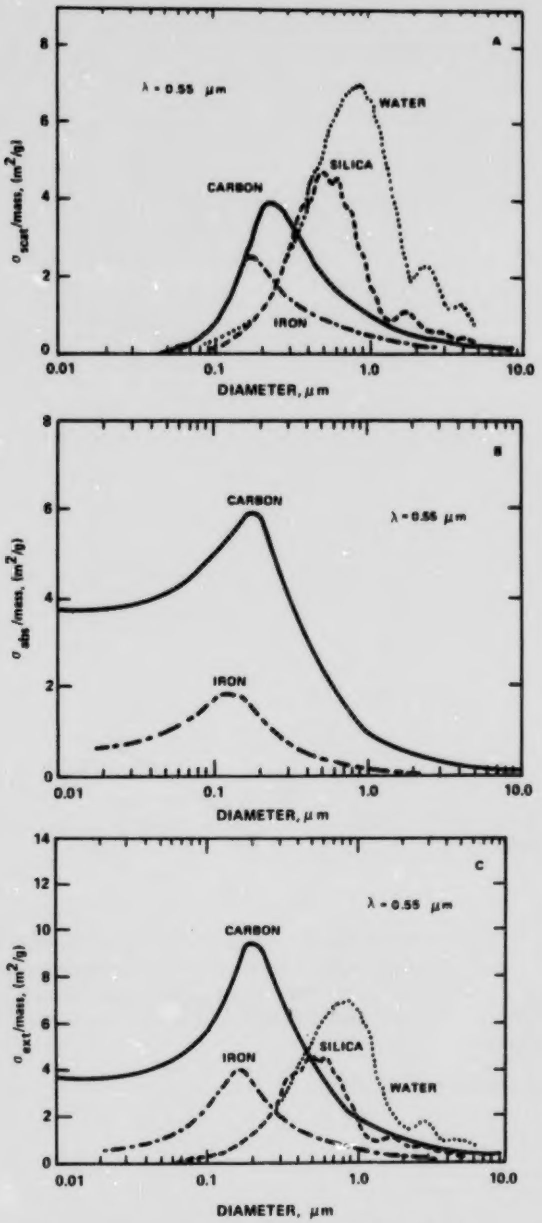


Figure 9-10. (A) Calculated scattering coefficient per unit mass concentration at a wavelength of 0.55 μm for absorbing and non-absorbing materials is shown as a function of diameter for single-sized particles. The following refractive indices and densities (g/cm^3) were used: carbon (m = 1.96-0.66i, d = 2.0), iron (m = 3.51-3.95i, d = 7.86), silica (m = 1.55, d = 2.66), and water (m = 1.33, d = 1.0). (B) Calculated absorption coefficient per unit mass concentration at 0.55 μm for single-sized particles of carbon and iron. (C) Calculated extinction coefficient per unit mass concentration at 0.55 μm for single-sized particles of carbon, iron, silica, and water.

Source: (a) Faxvog (1975); (b and c) Faxvog and Roessler (1978).

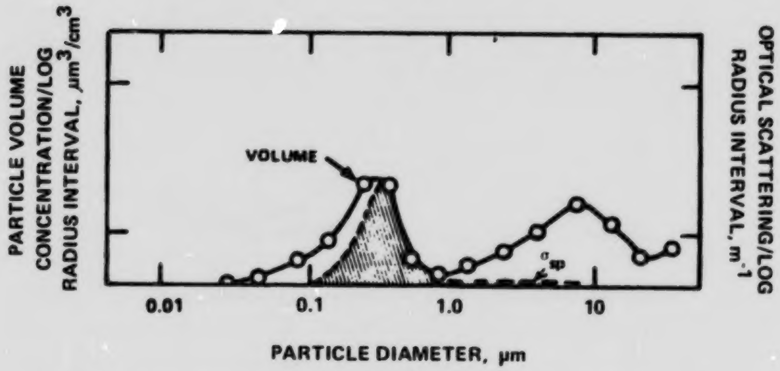


Figure 9-11. For a typical aerosol volume (mass) distribution, the calculated light-scattering coefficient is contributed almost entirely by the size range 0.1-1.0 μm. The total σ_{sp} and total aerosol volume concentration are proportional to the area under the respective curves.

Source: Charlson et al. (1978a).

ing one that matches the Eastern-Western, urban-nonurban grand average of Whitby and Sverdrup (1980). The calculated ratio for fine particles changes by no more than 40 percent as the MMD ranges from 0.2 to 1.0 µm (Figure 9-12). Figure 9-12 also demonstrates the relative inefficiency of coarse particles in degrading visibility.

The relative consistency of calculated light scattering per unit mass concentration over a range of particle distributions and the dominant influence of fine particles suggest that a reasonably good approximation of light-scattering coefficient can be obtained by measuring fine-particle mass concentration. Indeed, agreement from simultaneous monitoring of the two parameters at a wide variety of sites has been found by several investigators (Table 9-1). The "normalized" column of Table 9-1 normalizes the reported σ_{sp}/FMC values to a uniform nephelometer operating wavelength and the best nephelometer calibration values available (Ruby and Waggoner, 1981). With each study having equal weighting, the mean normalized σ_{sp}/FMC is 3.3 (±0.8) m^2/g; with each study weighted by the number of regression points, the mean normalized σ_{sp}/FMC is 3.2 (±0.8) m^2/g. Correlations between the fine-particle mass concentration and σ_{sp} are consistently high. Figure 9-13 (Macias and Husar, 1976) shows this relationship for St. Louis. It should be noted that in determining mass concentration gravimetrically (or by beta gauge), the filters are usually equilibrated at some reference RH before each weighing. Volatile components of the particles, such as water, may have desorbed (or adsorbed), thus changing the apparent mass concentration from its true atmospheric value. To compensate for this effect, many researchers heat the aerosol before nephelometry to reduce the RH and simulate filter equilibration. Not heating the aerosol before nephelometry can be expected to produce higher and more variable ratios of σ_{sp}/FMC. Heating the aerosol carries some risk of distorting the results because volatile components other than water (e.g., ammonium nitrate or organic particles) may also desorb. An unknown portion of the variation between studies in Table 9-1 is due to differences in: the volatility of the fine mass; the RH of filter equilibration; care in maintaining that humidity throughout the weighing procedure; the mass measurement method employed (gravimetric or beta gauge); the particle classifier employed (cutpoint and efficiency characteristics); the statistical procedures used; and the detailed calibration and operating conditions of the nephelometer used in the σ_{sp} determination. The remaining variation is of course due to actual differences of particle composition and size.

The high correlations indicate that, at the sites studied, fine-particle mass dominates particle scattering. This was documented in an experiment conducted by Patterson and Wagman (1977), who monitored the ambient aerosol size distribution with a set of four cascade impactors in the New York metropolitan area. One (background) impactor was operated only when the light-scattering coefficient was between 0 and 0.15 km^{-1}; impactor A at 0.15 to 0.3; impactor B at 0.3 to 0.45; and impactor C when values exceeded 0.45. The measured mass distributions (Figure 9-14; Patterson and Wagman, 1977) show that at good background visibility

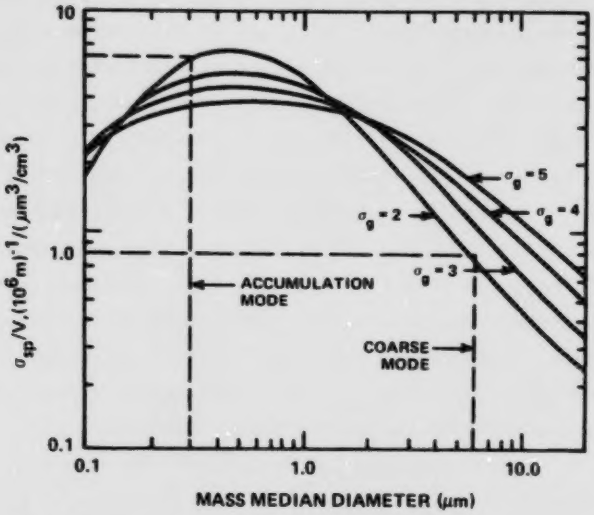


Figure 9-12. Scattering-to-volume concentration ratios are given for various size distributions. The ratio values for accumulation (fine) and coarse modes are shown by dashed lines corresponding to average empirical size distributions reported by Whitby and Sverdrup (1980).

Source: Latimer et al (1978).

TABLE 9-1. PARTICLE LIGHT SCATTERING COEFFICIENT PER UNIT FINE-MASS CONCENTRATION

Location	Reported σ_{sp}/FMC (m^2/g)	r	N	Normalized† σ_{sp}/FMC (m^2/g)	Effective Nephel. Wavelength* (nm)	Nephel. Calib.*	Nephel. Heated*	Classifier Cut Point* (μm)	Reference
Mesa Verde, CO	2.9	-	5	2.5	525	old	yes	2.5-3.0	1
Seattle, WA (residential)	3.1	0.95	58	2.9	550	old	yes	2.5-3.0	1
Seattle, WA (industrial)	3.2	0.97	64	3.0	550	old	yes	2.5-3.0	1
Puget Island, WA	3.0	0.97	26	2.8	550	old	yes	2.5-3.0	1
Portland, OR	3.2	0.95	108	3.0	550	old	yes	2.5-3.0	1
New York, NY	5.0	-	-	3.7	475	old	no	1.5	2
St. Louis, MO	5.0	0.96	60	4.4	525	old	?	3.0	3
Los Angeles, CA	3.7	0.83	39	2.8	475	old	no	~4.5	4
Oakland, CA	3.2	0.79	20	2.4	475	old	no	~4.5	4
Sacramento, CA	4.4	0.98	6	3.3	475	old	no	~4.5	4
Ohio (rural, winter)	3.5	0.85	55	3.1	525	old	yes	2.9	5
Ohio (rural, summer)	4.1	0.91	28	3.6	525	old	yes	3.0	6
Great Smoky Mtns., NC	3.4	0.97	69	3.0	525	old	yes	2.6	7
Shenandoah Valley, VA	4.2	0.96	36	4.4	538	new	yes	1.0	8
Houston, TX	4.4	0.96	14	4.6	538	new	yes	1.0	9
Denver, CO	3.4	0.97	208	2.5	475	old	yes	2.5	10
Shenandoah Valley, VA	5.8	0.91	59	5.0	475	new	yes	2.5	11,12

† Normalized to 525 nm and new calibration values according to Ruby and Waggoner (1981), assuming σ_{sp} proportional to $\lambda^{-1.5}$.

* Stated in or inferred from reference.

References: 1, Waggoner and Weiss (1980); 2, derived by Charlson et al. (1978a) from Patterson and Wagman (1977); 3, Macias and Husar (1976); 4, Samuels et al. (1973); 5, Nininger et al. (1981a); 6, Nininger et al. (1981b); 7, Ellenson et al. (1981); 8, Weiss et al. (1982); 9, Waggoner et al. (1982); 10, Groblicki et al. (1981); and 11, Ferman et al. (1981), as revised in 12, Wolff et al. (1982).

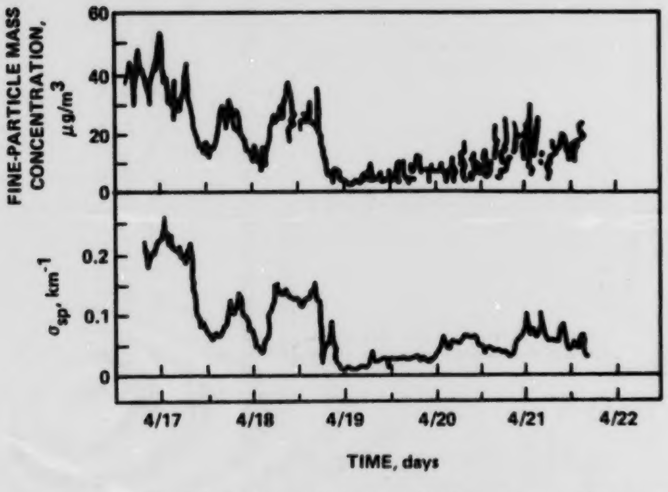


Figure 9-13. Simultaneous insitu monitoring of σ_{sp} and fine-particle mass concentration in St. Louis in April 1973 showed a high correlation coefficient of 0.96, indicating that σ_{sp} depends primarily on the fine-particle concentration.

Source: Macias and Husar (1976).

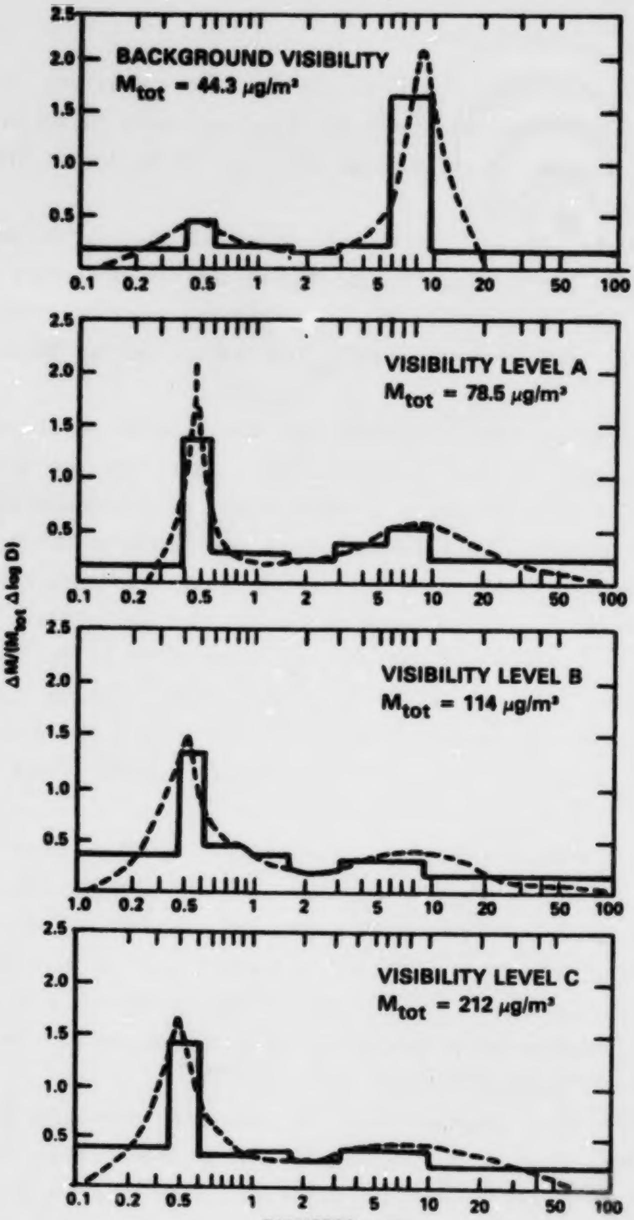


Figure 9-14. Aerosol mass distributions, normalized by the total mass, for New York aerosol at different levels of light-scattering coefficient show that at high background visibility, the fine-particle mass mode is small compared with the coarse-particle mode. At the low visibility level, C, 60 percent of the mass is due to fine particles. The solid lines represent histograms of the impactor data, while dashed lines represent the best fit of a bimodal, log-normal distribution.

Source: Patterson and Wagman (1977).

levels, coarse particles constituted 70 percent of the mass concentration. At the low visibility level C, however, over 60 percent of the total mass was contributed by fine particles. Thus, visibility in the New York metropolitan area was found to be lowest when the concentration of fine particles reached a maximum.

It is conceivable that in the arid Southwestern United States the aerosol refractive index and relative amounts of coarse and fine particles are so different that the σ_{sp}/FMC ratios quoted above would not be applicable. Preliminary results from project VISTTA, however, (Macias et al., 1980, 1981) suggest that σ_{sp}/FMC ratios in the Southwest are ~ 3 m^2/g as measured elsewhere.

In areas where fine-particle concentrations are low, coarse particles may contribute significantly to light extinction. However, coarse dust particles are much less efficient scatterers per unit mass (Figure 9-10), so that much higher mass concentrations are required to produce a given optical effect. In windblown dust, for example, Patterson and Gillette (1977) reported values of the ratio of light scattering to mass concentration that were more than an order of magnitude lower than those noted above for fine particles.

9.2.3.4 Particle Absorption--Particle absorption (σ_{ap}) appears to be on the order of 5 to 10 percent of particle extinction in remote areas such as a plateau outside Flagstaff, AZ (Weiss et al., 1979). Its contribution may rise to 50 percent of σ_{ext} in urban areas, with values of 10 to 25 percent typical for suburban and rural sites (Weiss et al., 1979; Waggoner et al., 1981; Groblicki et al., 1981).

The apparently large role of absorption in visibility in urban areas has prompted much research on absorption in recent years. Weiss et al. (1978) fractionated aerosol at many sites into coarse and fine portions (cut at 2.5 μm) and found that the fine particles accounted for 80 to 98 percent of absorption. They also determined the wavelength- (color-) dependence of σ_{ap} for the samples, which showed that at United States sites, σ_{ap} is approximately proportional to the inverse of wavelength. Thus σ_{ap} is capable of causing brownish coloration of objects viewed through an aerosol.

The extinction per unit mass concentration for absorbing aerosols is a figure of some importance because estimates show it to be significantly higher than for scattering-only aerosols. Figure 9-10(c) shows the theoretical value for carbon particles to be higher than for any other species considered (Faxvog and Roessler, 1978). Jennings and Pinnick (1980), relying on the small size of combustion particles (diameters generally less than 0.3 μm) and the approximate linearity of the extinction efficiency factor in this region, predicted a linear and size distribution-independent relation between extinction coefficient and mass concentration of carbon particles, with a ratio of 9.5 m^2/g at 0.55 μm. Laboratory studies by Roessler and Faxvog (1980) showed values of 9.8 m^2/g for acetylene smoke and 10.8 m^2/g for diesel exhaust. They also summarized results from other investigators on various aerosols that showed values of 6.1 to 9.5 m^2/g. In developing a spectrophone, Truex and Anderson (1979) measured an absorption of mass concentration ratio 17 m^2/g (at 0.417 μm wavelength) for aerosol from a propane-oxygen flame. As methods for measuring elemental carbon have improved, Groblicki et al. (1981) have performed atmospheric measurements of fine absorption/fine elemental carbon mass concentration in Denver and found an average of 11.8 m^2/g. While the amount of absorption per unit mass concentration depends on chemical composition and particle size distribution (Waggoner et al., 1973; Bergstrom 1973), the pattern emerging from these empirical and theoretical studies is that absorbing particles have a more significant visibility impact than their mass would indicate, probably by a factor of 3 to 4, compared to scattering-only particles.

Weiss and Waggoner (1981) calculated that, for constant mass concentration, changing 20 percent of each particle of a nonabsorbing aerosol to an equal volume of absorbing soot reduces visual range (or increases σ_{ext}) by about 35 percent. They pointed out the importance of this concept as fuel conservation practices (e.g., use of diesel engines, wood burning) lead to greater emissions of light-absorbing aerosol.

9.2.4 Chemical Composition of Atmospheric Particles

Given the dominance of particles in degrading visibility, it is natural that their composition should be studied. Such knowledge permits estimates of the roles various sources play in visibility impairment. This section discusses the commonly observed particulate species in the context of visibility impairment (see Chapter 5 for a more detailed discussion.) Before discussing the aerosol components currently believed to be of significance, it is important to consider some uncertainties in measurements and deductions.

Contrary to first impressions, visibility is an extremely complex subject, being determined by the sum of all atmospheric constituents, lighting conditions, the target, and the observer. These factors are all extremely variable. Although it is generally accepted that fine particles cause most visibility problems, the concentration and composition of these particles can vary considerably at different times and sites. Furthermore, field studies that characterize the aerosol must include diverse and sometimes elaborate instruments and techniques to measure all relevant parameters simultaneously. There is evidence of volatile aerosol (e.g., ammonium nitrate) that exists in the atmosphere and thus degrades visibility, but cannot always be retained by conventional filtration for subsequent analysis. Some fine particles (e.g., diesel exhaust) are distinctly nonspherical and hinder attempts to model their optical influence theoretically or even report a size distribution. Until recently, several fine mass components (e.g., ammonium nitrate and elemental C) have been measured either inaccurately or not at all. Water is a fine mass component that is particularly difficult to measure. Water degrades visibility only when in the liquid or solid phase, but it is ubiquitous and causes particles of several common species to grow or shrink, which can affect visibility drastically. Thick hazes or fogs are often dismissed as being caused solely by high humidity, whereas in some cases they might not have formed without the presence of anthropogenic hygroscopic nuclei.

Fortunately, there are reasons to believe we can circumvent these uncertainties. Improved analytical methods are evolving for nitrate measurement. Carbon measurement techniques are being intercompared and applied to the atmosphere. Empirical work by Janzen (1980) on highly irregular carbon black particles indicates that absorbing chain-aggregates (similar to diesel exhaust and flame soot), despite their extreme nonsphericity, can be modeled surprisingly well with Mie theory by assuming the aggregates to be spheres of equal volume. Relatively constant scatter/fine mass concentration ratios are being reported for a variety of sites (see Table 9-1). Extinction calculated from gross size distribution measurements is usually within a factor of two of measured values (Ensor et al., 1972; Patterson and Wagman, 1977). Regression analyses show consistent correlations between scattering or visibility and sulfate concentration (White and Roberts, 1977; Cass, 1979; Trijonis, 1978a,b).

Current knowledge indicates that fine aerosol is composed of varying amounts of sulfate, ammonium, and nitrate ions; elemental carbon and organic carbon compounds; water; and smaller amounts of soil dust, lead compounds, and trace species. The following discussion separates the components, although in reality they may exist as internal mixtures (i.e., coexist within the same particle).

Sulfate occurs predominantly in the fine mass (Stevens et al., 1978; Tanner et al., 1979; Lewis and Macias, 1980; Ellestad, 1980). Sulfate ion generally constitutes 30 to 50 percent of the fine mass at a wide variety of sites (Stevens et al., 1978; Pierson et al., 1980; Stevens et al., 1980; Lewis and Macias, 1980; Ellestad, 1980; Macias et al., 1981), although some urban sites have values of 10 to 20 percent, perhaps due to locally high values of other fine mass constituents (Countess et al., 1980b; Cooper and Watson, 1979). Sulfate usually occurs in combination with hydrogen and ammonium ions (Stevens, 1978; Pierson et al., 1980; Charlson et al., 1978b; Stevens et al., 1980; Tanner et al., 1979) and to a lesser extent calcium and magnesium. Indirect measurements of sulfate by examining the scattering response of atmospheric aerosol to changes of relative humidity confirm the prevalence of H_2SO_4 and its ammonium salts (Weiss et al., 1977; Waggoner et al., 1981). Regression analyses by Cass (1979), White and Roberts (1977), Trijonis and Yuan, (1978a,b), Grosjean et al. (1976), Leaderer et al. (1978), and Heisler et al. (1980) show significant correlations between sulfate concentrations and visibility or extinction. High correlations between variables do not necessarily imply cause-and-effect; however, a lack of correlation would imply the absence of cause-and-effect.

Ammonium ion is typically found to account for 5 to 15 percent of the fine mass (Lewis and Macias, 1980; Patterson and Wagman, 1977; Countess et al., 1980b). It often correlates well with sulfate levels (Tanner et al., 1979). Because of the possible reaction of ammonia with previously collected acidic particles, reported ammonium ion values may be higher than actually exist in the atmosphere.

Recognized sampling problems prevent valid statements about ambient particulate nitrate concentrations at present (Appel et al., 1979; Spicer and Schumacher, 1977 and 1979). Simple filtration (even with nonalkaline, nonreactive, high-purity filters) may not give true values, due to the tendency of ammonium nitrate to seek equilibrium with ammonia and gaseous nitric acid during sampling and storage.

Concentrations of apparent elemental carbon have been reported by Wolff et al. (1980) for a variety of United States sites. Apparent elemental carbon was found to range from 1.1 $\mu g/m^3$ at a remote site to an average of 5.9 $\mu g/m^3$ for urban sites, with about 80 percent in the fine fraction. The word "apparent" is used because the value may be high due to charring of some organic particles during the organic analysis. Shah et al. (1982), employing a combustion technique that included a quantitative correction for charring based on reflectance change, has analyzed archived NASN hi-vol filters collected during 1975 at 46 urban and 20 rural United States sites. The mean annual elemental carbon concentration was 3.8 $\mu g/m^3$ at the urban sites and 1.3 $\mu g/m^3$ at the rural sites. Countess et al. (1980b) determined that elemental carbon accounted for 15 percent of Denver's fine mass. Considering the high extinction efficiency of elemental C reviewed in Section 9.2.3.4, elemental C obviously has a significant effect on Denver's visibility. Several investigators have concluded that elemental carbon is the only significant light-absorbing species, including Rosen et al. (1978) who employed Raman spectroscopy, Allegrini (1980) who examined the spectral dispersion of the imaginary part of the refractive index, and Pierson and Russell (1979) and Weiss et al. (1979) who employed various solvent extraction schemes.

Determinations of particulate organic carbon concentrations are not discussed because of uncertainties in their measurement from adsorption or volatilization. Also, most reported organic particle concentration data are for unfractionated samples (i.e., all particle sizes present). Therefore, even though some sampling and analytical techniques may have validity, the interpretation of the concentrations for visibility purposes is hindered by the possible contribution of coarse organic particles. Improved techniques for organic particulate measurement are being developed.

Minor contributions to fine mass are made by soil-related elements, lead compounds (especially in urban areas), and trace species (Stevens et al., 1978).

It is suspected that soil particles significantly impair visibility mostly in arid or semiarid areas (Patterson, 1977) (in the United States, the Southwestern states). This observation may be due more to the relatively low fine-particle concentrations there, than to high concentrations of soil particles. Macias et al. (1981a) indicated that coarse particles accounted for 6 to 24 percent of extinction at Zilnez Mesa, Arizona, during the 3 days reported from summer 1979 field work. What fraction of coarse particles is derived from natural sources has not been established. Hall (1981) points out that winds and dust devils probably entrain much more dust when over anthropogenically disturbed soil surfaces (e.g., unpaved roads, off-road-vehicle trails) than when over undisturbed soils. Soil dust, whether stirred up by winds or by anthropogenic turbulence, almost always forms coarse aerosol (particle diameters greater than about 1 to 3 μm), although a few of the softer species such as carbonates can form submicrometer particles (Draftz, 1974).

9.2.4.1 Role of Water in Visibility Impairment--Natural fluctuations of RH can greatly influence the extinction of light by an aerosol. Since RH generally increases following sunset due to declining temperature (assuming a constant dew point), particles usually grow after sunset. After sunrise relative humidity usually declines as temperature rises, causing particles to shrink as they release water to the vapor phase.

As mentioned earlier, water affects visibility only when in the liquid or solid phase. Unfortunately, direct measurement of liquid water's contribution to mass is difficult due to its rapid phase change and the fact that, except in fogs, typically less than 0.01 percent of all water in a given volume exists in the liquid phase. For example, at 20°C, sea level, and 50 percent RH, the water vapor concentration is 8.65 g/m^3. By comparison, an aerosol composed of 0.5-μm diameter water droplets of sufficient concentration that $\sigma_{sp} = 0.2\ km^{-1}$, has a liquid water concentration of 42 $\mu g/m^3$. Ho et al. (1974) report a method employing microwaves specific to unbound liquid water. Applying it to the Los Angeles Basin, they found that for relative humidities between 40 and 70 percent water comprised 5 to 30 percent of the aerosol mass. Data from Mainz presented by Hänel (1976) are in good agreement. It has been known for many years that relative humidities above about 70 percent will often greatly reduce visibility due to the size growth of common aerosol species such as ammonium sulfate and sea salt (Orr et al., 1958).

A straightforward demonstration of the effect of RH on light scattering is seen by measuring the scattering coefficient of an atmospheric aerosol while increasing its RH (Covert et al., 1972). Figure 9-15 shows typical humidograms observed around the United States. The method cycles the aerosol's RH in a short period, so the input aerosol can be assumed not to have changed in composition or concentration while being measured.

Particles of certain inorganic salts commonly observed in the atmosphere (e.g., ammonium sulfate and sodium chloride) exhibit the phenomenon of deliquescence (i.e., an abrupt transformation from solid particle to liquid droplet, and growth at a RH specific to each compound). Above the deliquescent RH, the droplets absorb water and grow smoothly as RH increases (Orr et al., 1958; Charlson et al., 1978b; Tang, 1980). As RH decreases, salt particles that have already deliquesced do not crystallize until a RH well below the deliquescent RH is achieved (Tang, 1980; Orr et al., 1958), a phenomenon generally referred to as hysteresis. Until crystallization occurs, droplets become supersaturated, lose water gradually, and shrink

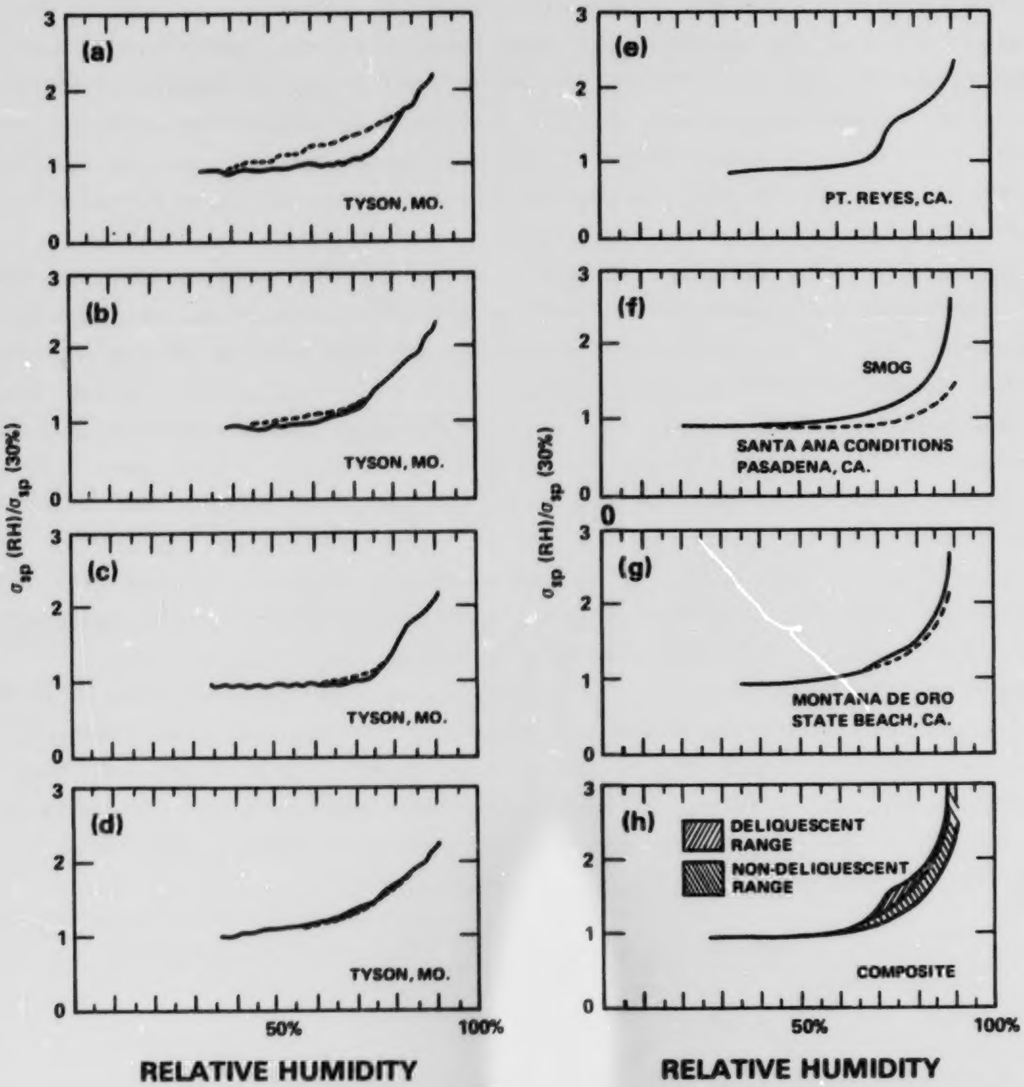


Figure 9-15. Humidograms for a number of sites show the increase in σ_{sp} which can be expected at elevated humidities for specific sites or aerosol types (marine, Point Reyes, CA; sulfate, Tyson, MO) and the range observed for a variety of urban and rural sites (composite). Dashed lines in a-d show the humidogram without injection of ammonia vapor, whereas the solid lines represent those runs with ammonia injection. In f, the different lines differentiate between smog and Santa Ana conditions, neither with any ammonia injection. In g and h, the range of humidograms is shown.

Source: Covert et al. (1980).

gradually as RH declines. If crystallization has not occurred and RH increases, the growth of droplets will follow the smoothly hygroscopic curve along which it just fell. Figure 9-16 depicts the behavior of an ammonium sulfate particle.

Hysteresis explains the persistence of some hazes at relative humidities below that at which they formed (Orr et al., 1958). The RH of crystallization depends on the size of insoluble nuclei present in each droplet. The dissipation of a water-enhanced haze may not occur abruptly as RH falls (even though it may have formed abruptly) because the sizes of nuclei within the droplets may vary, causing the droplets to crystallize at different relative humidities.

The hygroscopic salts discussed above do not necessarily exist as external mixtures (i.e., as pure and separate particles), but may be internally mixed (i.e., coexist within the same particle). Tang et al. (1978) have shown that the phase diagram of a multi-component system allows accurate prediction of its phase transformation and growth. In the important sulfuric acid-ammonium sulfate system they found the molecular composition to be of considerable importance. For example, for ammonium sulfate/total sulfate ratios above 0.95, deliquescence occurs at 80 percent RH, for ratios 0.75 to 0.95 at 69 percent, for ratios 0.5 to 0.75 at 39 percent, and for a ratio below 0.5 growth is not deliquescent but smooth. In some cases (e.g., letovicite, $(NH_4)_3\ H(SO_4)_2$) the phase diagram predicts two-stage growth, which was confirmed by experiment. The phase diagram approach apparently leads to quite accurate predictions of the growth of pure and mixed deliquescent aerosol particles.

To avoid the stringent requirement of knowing the detailed internal composition of the particles, other investigators have pursued theoretical and empirical simplifications. The most extensively developed model of this type is that of Hänel (1971, 1976). His model predicts that the extinction efficiency factor remains almost constant below relative humidities of 90 percent due to the rise in size parameter from particle growth and simultaneous reduction of refractive index from dilution by water. Thus,

$$\frac{\sigma_{sp}\ (RH_2)}{\sigma_{sp}\ (RH_1)} \approx \left[\frac{r\ (RH_2)}{r\ (RH_1)}\right]^2 \qquad (9\text{-}13)$$

where r = particle radius.

Using a semiempirical formula for the size growth of particles, he arrives at:

$$\frac{\sigma_{sp}\ (RH_2)}{\sigma_{sp}\ (RH_1)} \approx \left[\frac{1 - RH_1}{1 - RH_2}\right]^{2\varepsilon} \qquad (9\text{-}14)$$

where ε = constant ($\approx$ 0.25.) This result is appealing in its simplicity and its independence of size distribution and composition. Several restrictions, however, apply:

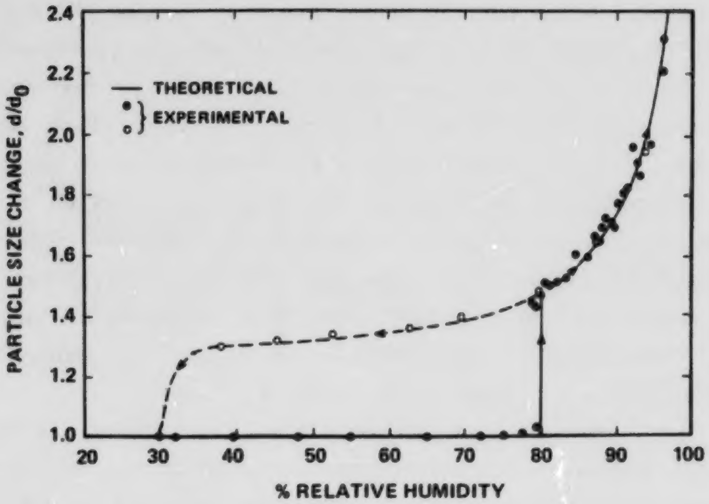


Figure 9-16. Relative size growth as a function of relative humidity for an ammonium sulfate particle at 25° C.

Source: Tang (1980).

1. Particle composition is the same for all sizes;

2. The particle diameters of interest for visibility are 0.2 to 2 μm;

3. Applicable only for $RH_2 < 95$ percent and RH_1 of 40-80 percent, unless one assumes a model aerosol and changes ε;

4. Inapplicable to maritime aerosol that has not increased from very low RH.

The first restriction causes some concern. It is not clear that fine particles have uniform composition. Though absolute nitrate concentrations are suspect, both Appel et al. (1978) and Patterson and Wagman (1977) indicate that sulfate and nitrate may possess different MMD's. Countess et al. (1980a) found that in Denver the mean daytime MMD's of sulfate and nitrate were 0.36 and 0.30 μm, respectively. Any success of Hänel's theory must be due to the frequent atmospheric observation that ammonium sulfate is the predominant hygroscopic species. The first and second restrictions may be severe in cities, where absorption by elemental carbon is significant because elemental carbon resides mostly below 0.2 μm diameter and is unlikely to be as hygroscopic as sulfate. The fourth restriction is necessary because of hysteresis. Even for nonmaritime aerosol, ambient RH often does not become small enough to cause crystallization of all major sulfate and nitrate species; thus, this restriction may be serious. Atmospheric testing of Hänel's theory is difficult as suggested by Hänel (1976) and demonstrated by Presle and Pirich (1980) due to continual advection and the resulting potential changes of particle concentration, size, and type.

In summary, the role of water in visibility impairment can be significant and is difficult to predict accurately to better than a factor of two, unless detailed and accurate data are available about the hygroscopic particles from 0.1 to 2.0 μm diameter. Natural fluctuations of RH will continue to occur. Their influence on visibility depends on the presence of hygroscopic particles, both natural and anthropogenic.

9.2.4.2 Light Extinction Budgets--Light extinction budgets (LEBs) are an attempt to assign a percentage of the total extinction to each chemical species. A common mistake in assessing the compounds responsible for visibility impairment is to assume that a compound's contribution to mass or fine mass is proportional to its contribution to visibility impairment. Its visibility contribution can be greater or lesser than its mass contribution, depending primarily on its size distribution and refractive index. The great usefulness of LEB's is to exonerate or implicate emission sources as primary causes of visibility impairment. If impairment can be demonstrated to be due mostly to compound A, and source X emits no compound A (or its precursors), source X probably has little to do with the visibility problem. The complex issues of pollutant dispersion and transformation are thus avoided. Reasonably complete LEB's have been possible only recently because of a lack of good data for key species (e.g., sampling artifacts for nitrates and organics, imperfect instrument performance, no technique for elemental carbon). Another problem is their expense. Many diverse and sometimes elaborate techniques must be applied to measure all relevant parameters simultaneously for a statistically significant period.

Two approaches have been used to arrive at light extinction budgets: (1) measurement of each species' size distribution and calculation of σ_{ext} by Mie theory; (2) statistical analysis, usually multiple linear regression. The former method requires detailed size distributions for each species, which is a problem because many significant species have MMD's at the steepest part of the extinction efficiency curve (Figure 9-7). Thus, a small error in the size distribution measurement may cause a large error in the calculated extinction. Further, detection limits, artifact formation, volatilization, unknown particle density, or imperfect instrument performance often lead to substantial uncertainties in the size distribution. The advantages of the first approach are that it is not affected by interdependencies among pollutant concentrations and that it allows an independent estimation of σ_{ext}, which can then be compared to the observed σ_{ext}.

Using multiple linear regression, one calculates a coefficient for each species that is then multiplied by that species' concentration to yield its contribution. Calculation of the coefficients requires use of the observed σ_{ext}, so this method does not allow an independent check of the results. Multiple linear regression may produce misleading results when the independent variables are highly correlated or when some variables are omitted (Snedecor and Cochran, 1980). In applying this technique to determine a LEB it is important that only the fine particles of the species be included in that species' concentration measurement; otherwise the coarse portion (relatively unimportant in visibility) will degrade the accuracy of the LEB.

Regrettably, few field studies have produced reasonably complete LEB's. Most older studies did not include key species or did not use only the fine component of species. Other studies have been of very short term and may be of limited statistical significance. Two studies will be presented herein, but it must be recognized that they may not be representative of other sites or even of the same sites at different times.

The most complete urban LEB to date is that for Denver in the winter (Groblicki et al., 1981). Data were collected for 41 days during November and December 1978. Employing multiple regression analysis, they found that, on the average, elemental carbon accounted for 38 percent of the variable part of extinction (i.e., all extinction except Rayleigh scattering extinction), ammonium sulfate 20 percent, ammonium nitrate 17 percent, organic compounds 13 percent, other PM 7 percent, and NO_2 6 percent.

The second example is for Zilnez Mesa, Arizona. LEBs' were calculated for 3 days in the summer of 1979, using the size distribution-Mie approach (Macias et al., 1981; Ouimette, 1980). For these few days, σ_{ext} caused by particles averaged 33 percent organic carbon, 18 percent ammonium sulfate, 13 percent elemental C, 22 percent other fine species, and 15 percent coarse particles.

No matter how accurately they may eventually be assessed, light extinction budgets are predictions only for the present conditions of sources, meteorology, etc. Because of the physical/chemical interactions among fine particles in the atmosphere, a good light extinction budget may prove untenable as conditions change.

9.2.5 Considerations in Establishing a Quantitative Relationship Between Fine-Particle Mass Concentration and Visual Range

For consistency, visual range must be defined in terms of a black target. Otherwise, the same atmospheric conditions will allow different values of visual range depending on the intrinsic contrasts of different targets, a confusing and misleading situation that must be avoided. For this reason, the use of airport visibility observations must be treated with caution. Airport visibility markers are defined as "Dark or nearly dark objects viewed against the horizon sky during the day," however, official guidelines for selecting the marker set specify that "Insofar as possible, markers of the type described in paragraph 2.7 should be used for determining visibility." (National Weather Service, 1979). There are no defined quantitative selection criteria, nor even rules that the chosen markers be documented quantitatively as to intrinsic brightness. Airport visibilities may be useful for visibility trend analysis, presuming that careful checks are made for marker and observer changes, but comparisons among airports or with instrumental determinations may be fraught with uncertainties.

Beginning with Equation 9-5 and the (usually) reasonable assumptions made in deriving it,

$$V = \frac{-\log_e \varepsilon}{\sigma_{ext}} = \frac{K}{\sigma_{ext}} \qquad (9\text{-}15)$$

where V = visual range

ε = observer's contrast threshold

σ_{ext} = extinction coefficient.

Letting $\gamma = (\sigma_{sp} + \sigma_{ap})/\mathrm{FMC}$, (FMC represents fine mass concentration),

$$\mathrm{FMC} = \frac{K}{\gamma V} - \frac{\sigma_{sg} + \sigma_{ag}}{\gamma} \qquad (9\text{-}16)$$

where σ_{sg} = scattering by gases (Rayleigh scattering) and σ_{ag} = absorption by gases.

There are thus two unknowns, K and γ, which must be established to relate FMC and V, assuming that extinction by gases is small. It is, of course, possible to relate FMC and V directly; however, studies have historically reported either K or γ. Their separate determination distinguishes the effects of contrast threshold (on K) from those of RH (on γ). We thus have a better physical understanding of why variations exist at different sites and times.

As discussed at the beginning of Section 9.2, no fixed value of K is generally applicable because observer contrast threshold varies with many factors and because it may be desirable to decrease K if greater than 50-percent detection probability is desired.

The choice of γ requires attention to several factors, most notably the effect of RH. γ is usually determined empirically from a ratio of a measurement of σ_{ext} to FMC determined from filters equilibrated at a medium RH (typically 50 percent); however, the σ_{ext} determination often uses a nephelometer operated at an RH substantially different from ambient RH (either intentionally or unintentionally). Thus, using these γ values with FMC from equilibrated filters results not in an estimate of in situ σ_{ext}, but in an estimate of σ_{ext} at a different RH. To correct for the RH effect, it is usually necessary to increase γ by a factor of about 1.5 at 80-percent RH, about 2 to 3 at 90-percent RH, etc. (see Figure 9-15). The frequency of occurrence of high RH is of obvious importance in choosing the factor. Table 9-2 addresses this point. For example, in January at 6 a.m., half the stations report RH $\geq$ 50%, 98 percent of the time. By midafternoon, half of the stations report RH $\geq$ 50 percent only 71 percent of the time. Though high ($\geq$ 90 percent) daytime RH occurs relatively infrequently, the effect of hysteresis must be remembered: until a RH substantially lower than the deliquescent RH is reached, deliquescent particles (e.g., ammonium sulfate) remain as droplets and continue enhanced scattering due to water. The large daily variation of RH at a given site, hysteresis, and sites where hygroscopic particles may not dominate fine mass complicate the choice of a fixed factor for accounting for RH effects on γ.

Other factors that influence γ include the size and composition of fine particles. One expects that at sites with high fine mass concentrations the fine particles will grow to somewhat larger sizes than at relatively cleaner sites, due to increased coagulation rates. Because of the steepness and monotonicity of the extinction efficiency curve in the usual fine-particle diameter range, γ will be enhanced in high loading areas relative to cleaner sites; however, the enhancement factor is quite dependent on the size distributions involved. The effect is no doubt of second order compared to RH effects on γ.

Fine particle composition may have two effects on γ. First, in areas with significant absorption by particles (e.g., urban and industrial sites), the σ_{sp}/FMC values reported in Table 9-1 are underestimates of γ, perhaps by 10 to 50 percent. Second, if the volatile fraction of fine mass changes from that present during each of the Table 9-1 studies, γ should be

TABLE 9-2. MEDIAN PERCENT FREQUENCY OF OCCURRENCE OF SELECTED RH CLASSES FOR 54 STATIONS IN THE CONTIGUOUS U.S.

Time[a]		Relative Humidity		
		≥ 50%	≥ 70%	≥ 90%
January,	6 a.m.	98	82	29
	3 p.m.	71	35	10
April,	5 a.m.	97	77	27
	3 p.m.	44	18	4
July,	4 a.m.	100	95	36
	3 p.m.	46	12	2
October,	5 a.m.	98	84	37
	3 p.m.	42	14	4
Annual average	a.m.	98	85	32
	p.m.	51	20	5

Source: Derived from U.S. Department of Commerce (1968). Based mostly on the years 1949-1954.

[a] Average for each midseasonal month. Hours (in local standard time) were selected to conform most nearly to the average national daily maximum (early morning) and minimum (3 p.m.) RH.

increased if the volatile fraction increases and decreased if the volatile fraction decreases. As noted earlier, the detailed sampling and storage conditions for γ determination may influence the result significantly when volatile species are present. That atmospheric concentrations of volatile species may change can be predicted from projected emissions and atmospheric chemistry. For example, current estimates of emission trends in the Ohio River Basin from the mid-1970's to the year 2000 are that SO_2 emissions will increase only slightly, while NO_x emissions will grow substantially (Glass, 1978; Stukel and Keenan, 1977). Because a portion of NO_x emissions eventually form volatile nitrate particles (Orel and Seinfeld, 1977), there may be increases in the volatility of fine mass downwind of this region in the coming years. Present developments in the measurement of volatile compounds will reduce our uncertainty about this aspect of γ as they come into use.

The net effect of all these factors (except a decrease in volatility) is to increase the value of γ from σ_{sp}/FMC values reported in Table 9-1. The points discussed above should be considered in choosing a fixed value for γ.

Figures 9-17 and 9-18 show the relationship of fine mass concentration to K, γ, and visual range, in one graph for a fixed visual range and in the other for a fixed K. Note that these figures deal with visual range, which airport visibility observations always underestimate to some extent. In both figures Rayleigh scattering is taken into account but σ_{ag} is assumed to be zero.

9.3 VISIBILITY AND PERCEPTION

The term "visibility" is used colloquially to refer to various characteristics of the optical environment, such as the clarity with which distant details can be resolved and the fidelity of their apparent coloration.

Traditionally, visibility has been defined in terms of visual range: the distance from an object that corresponds to a minimum or threshold contrast between that object and some appropriate background. Threshold contrast refers to the smallest brightness difference between two stimuli that the human eye can distinguish. The measurement of these quantities depends on the nature of the observer, his or her physical health, and state of attention or distraction, among other things.

Although visibility defined by visual range is a reasonably precise definition, visibility is more than being able to see a target at a distance for which the contrast is reduced to the threshold value. Visibility also includes seeing vistas at shorter distances and being able to appreciate the details of line, texture, color, and form. Any definition of visibility and the selection of methods for monitoring visibility impairment should relate to these

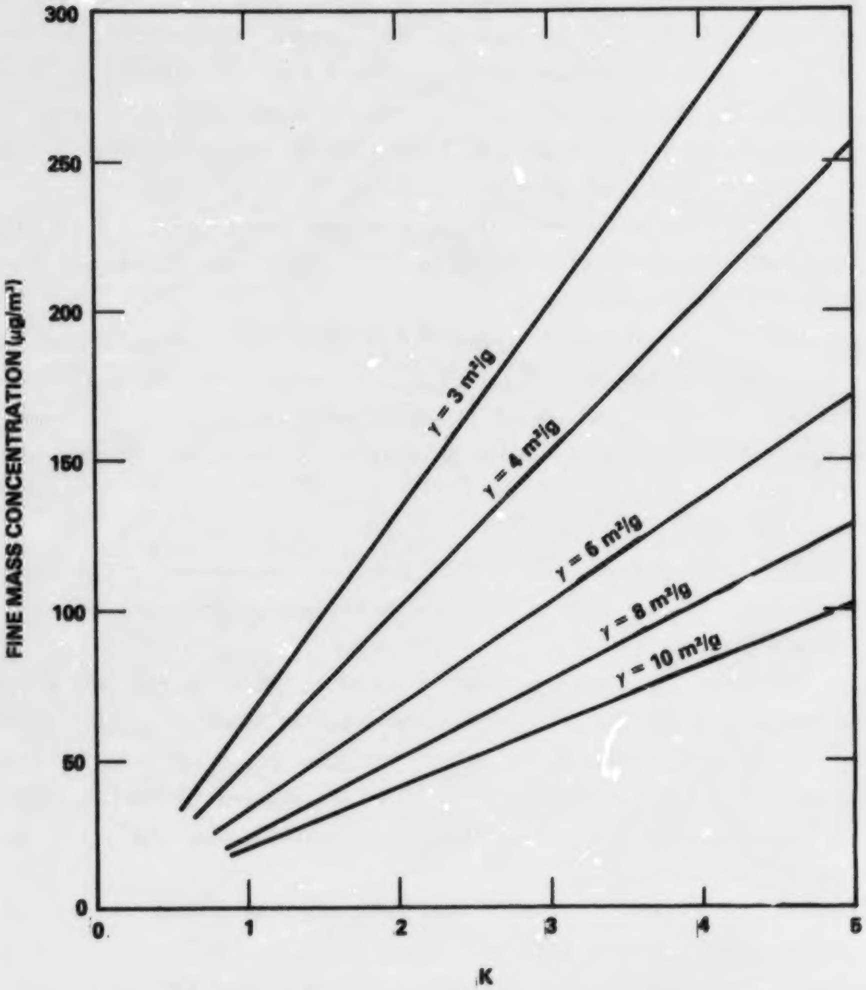


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Figure 9-17. Fine mass concentration (determined from equilibrated filter) corresponding to 4.8 km visual range, as a function of K and γ, where K equals the Koschmieder constant ($-\log_e \varepsilon$), and γ equals ($\sigma_{sp} + \sigma_{ap}$)/fine mass concentration.

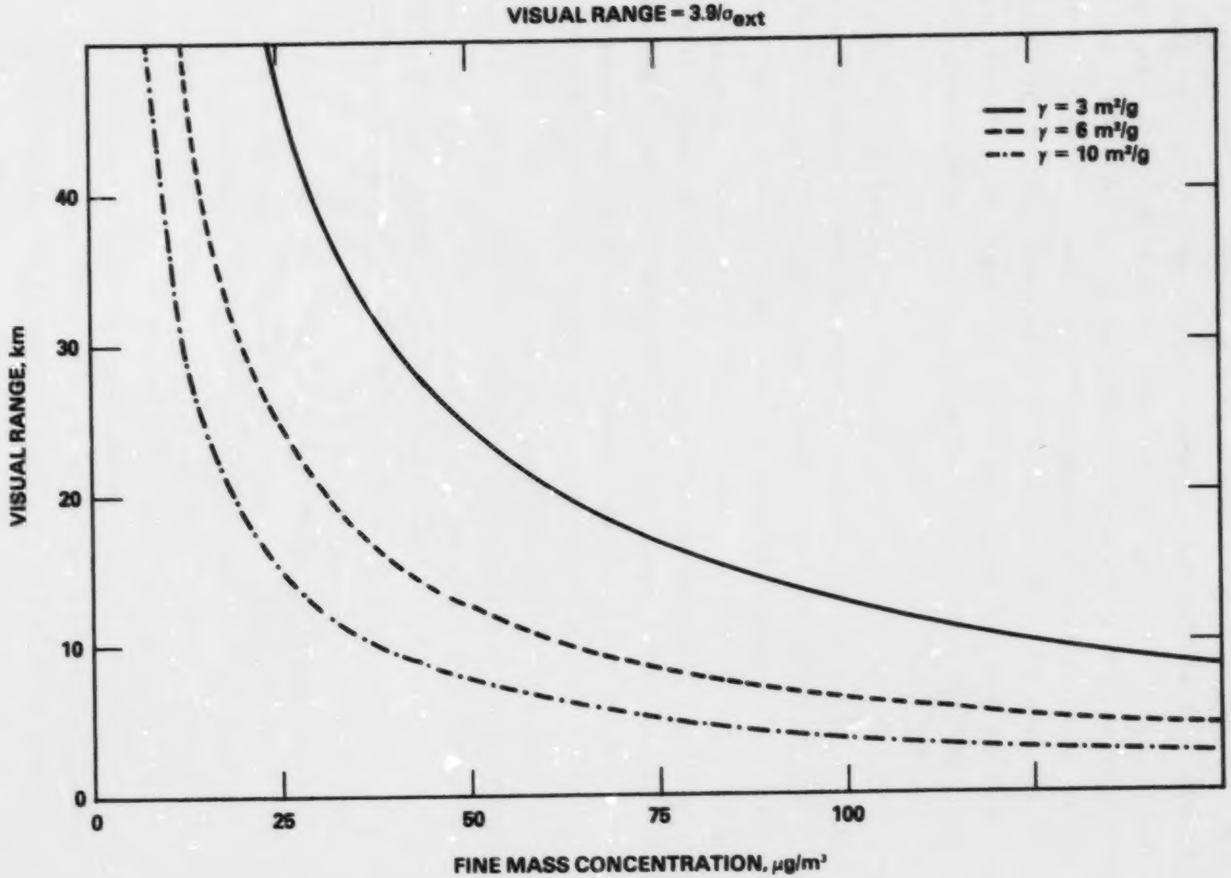


Figure 9-18. Visual range as a function of fine mass concentration (determined from equilibrated filter) and γ, assuming K=3.9.

aspects of perceiving distant objects. The selection of a good parameter (or parameters) to characterize visibility depends on whether one's interest is in human perception of visual air quality or in the cause of visibility degradation (that is, the air pollution itself). Apparent target contrast relates well to how a person perceives visual air quality and serves as the fundamental measure of visibility impairment (U.S. Environmental Protection Agency, 1979). Color change is also a useful measure. Fine particle concentrations and scattering coefficients are good measures when relating visibility impairment to pollutant sources. Visual range cannot be measured directly by an instrument. Visual range, however, can be derived from instrumental data.

Visual impairment resulting from air pollution can occur as layered haze or as uniform haze (Malm et al., 1980a,b,c). Layered haze produces a visible spectral discontinuity between itself and background (sky or landscape) while uniform haze exhibits reduced overall air clarity. The classic example of a layered haze is a tight, vertically constrained, coherent plume. As the atmosphere changes, however, from a stable to an unstable condition and a plume mixes with the surrounding atmosphere, the diffused plume, though no longer visible as a layer of haze, may reduce overall air clarity. The perceived effects of layered and diffused haze are quite different, because the eye is much more sensitive to a sharp demarcation in color or brightness than it is to a gradual change over time (Green, 1965; Patel, 1966). Since a change in uniform haze takes place over hours or days, an evaluation of visual air quality change resulting from a uniform haze requires remembering what the scene looked like before a change in air pollution took place. A layered haze, on the other hand, is evaluated by direct comparison with the background. Whether the pollution occurs as layered or uniform haze, judgments of visual air quality as a function of air pollution might be altered by variations in sun angle, cloud cover, and/or landscape features.

An assessment of visibility impairment produced by air pollution must consider the process of human visual perception, as well as changes in the optical characteristics of the atmosphere. The perception of brightness, contrast, and color is not determined simply by the pattern and intensity of incoming radiation; rather, it is a dynamic searching for the best interpretation of the visible scene. The relative brightness of an object may vary as a function of its background, even though its absolute brightness remains constant. For example, consider the difference between a candle in a brightly lit room and one in a dimly lit room. The contrast between an object and its surroundings is fundamental to visibility. As the contrast between object and background is reduced (for example, by fog or air pollution), the object becomes less distinct. When the contrast becomes very low, the object will no longer be visible. This liminal or threshold contrast has been the object of considerable study. The threshold contrast is of particular interest for atmospheric visibility, since it influences the maximum distances at which various components of a scene can be discerned. Equally important to visibility is the smallest perceptible change in contrast in a viewed scene caused by an increase in pollution haze (U.S. Environmental Protection Agency, 1979).

Malm et al. (1980a) investigated the relationship of contrast and color, and of changes in these variables, to the perception of visual air quality. They determined that the various demographic backgrounds of visitors to a national park did not influence perception, but that changes in color contrast did influence the accuracy and consistency of perception of visual air quality. In photographic slides of a mountain scene used as the test vehicle, color contrast was determined by variables such as weather condition, time of day, and ground cover, as well as by amount of air pollution. Although an incremental color contrast change was perceived to be the same across air pollution levels, clean air environments appeared to be more sensitive to contrast changes. The evidence indicates that a change in air pollution level produces a larger contrast change in clean air than in relatively dirty air and is, therefore, more perceptible.

For small objects, the size of the visual image on the retina of the eye also plays an important role in the perception of contrast. As an object recedes from us and apparently becomes smaller, details with low contrast become difficult to perceive. The reason for this loss of contrast perception is not only that the relative brightness of adjacent areas changes, but also that the visual system is less sensitive to contrast when the spacing of contrasting areas decreases. If the contrast spacing is a regular pattern of light and dark bands, (e.g., a picket fence), a "spatial frequency" can be readily described by the number of pattern repetitions or "cycles" per degree of viewing angle. The human eye is much more sensitive to contrast at certain spatial frequencies than at others (cited in U.S. Environmental Protection Agency, 1979).

The relationship between perceived contrast threshold and target characteristics (size and pattern) is important for visibility because a scenic vista usually contains a number of targets of varying sizes and arrangement. The calculation of the perceptibility of all targets requires specification of their angular size distribution. The perception of "texture," consisting of contours of small angular size and high spatial frequency, is particularly affected by this loss of threshold sensitivity.

Measurements of pattern perception thresholds by several researchers (Van Nes and Bouman, 1967; Schober and Hilz, 1965; Cornsweet, 1970) make it clear that the scattering and absorption of light by particles and gases added to the atmosphere can lead to a dramatic loss of visibility through contrast reduction. The operator of a motor vehicle or the pilot of an airplane who must react quickly to minimal visual cues may be greatly disadvantaged by an increase in atmospheric pollutants. This loss in contrast could make a target that is normally visible at 100 meters (109 yards) visible only at 20 meters (22 yards), although a reduction in visibility to levels this low rarely occurs.

In many pristine areas, where viewing distances are 50 to 100 kilometers, the most readily apparent effect of incremental pollution is a reduction in apparent contrast and discoloration of nearby objects and sky. For large targets, calculation of contrast changes accompanying increasing particle levels indicates that the maximum decrease in contrast occurs for objects located at distances of about one-fourth of the visual range from the observer (Malm, 1979a,b). Thus, in an initially clean atmosphere, an increase in fine particles produces maximum contrast reduction for large objects 50 to 100 kilometers away. A reduction in visual range of 5 percent results in a reduction in contrast of 0.02 for those objects. Such a change may be just perceptible. The contrast detail (texture, small objects) and coloration of closer objects may, however, be affected to a greater degree (Henry, 1979; Malm, 1979a,b).

The perceived color of objects and sky is also changed by the introduction of fine particle aerosols. Because it is difficult to specify perceived color, only a qualitative description is possible. In general, as distance from the observer increase, the apparent color of a target fades toward the hue of the horizon sky. Without particles, scattered air light is blue, and dark objects appear increasingly blue with distance. The addition of small amounts (1 to 5 $\mu g/m^3$) of fine particles throughout the viewing distance tends to whiten the horizon sky, making distant dark objects and the intervening air light (haze) appear more gray. According to Charlson et al. (1978a), even though the visual range may be decreased only slightly from the limit imposed by Rayleigh scattering, the change from blue to gray is an easily perceived discoloration. The apparent color of white objects is less sensitive to incremental fine particle loadings. Increments in particle levels produce a much greater color shift in cleaner atmospheres (Malm, 1979a,b).

Aerosol haze can also degrade the view of the night sky. Star brightness is diminished by light scattering and absorption. Perception of stars is also reduced by an increase in the brightness of the night sky caused by scattering of available light. In or near urban areas, night sky brightness is significantly increased by particle scattering of artificial light. The combination of extinction of starlight and increased sky brightness markedly decreases the number of stars visible in the night sky at fine particle concentrations of 10 to 30 $\mu g/m^3$ (Leonard et al., 1977).

Thus, aerosol haze reduces visual range and contrast, and changes color perception. Visually, the objects are "washed out" and the aesthetic value of the vista is degraded, even though the distances are small relative to the visual range.

Although natural sources of light scattering and light absorbing aerosols are undoubtedly important in producing geographical and seasonal patterns of visibility impairment, analysis of visibility trends and other information discussed in later sections suggests that manmade air pollution is a significant influence. It is also important to note that changes in pollution levels are most easily perceived in regions with the best visibility.

9.4 HISTORICAL PATTERNS OF VISIBILITY

Records of visual range can be used to gain insight into the effects of changing emission patterns on visibility. As an example, Marians and Trijonis (1979) have derived statistical relationships between light extinction (computed from visibility data) and historical emission trends. Yearly values of extinction from four Arizona airports were regressed against statewide emissions of smelter SO_x, nonsmelter SO_x, NO_x, and RHC (reactive hydrocarbons). Smelter SO_x was found to be the most significant variable. Particularly close relationships between Arizona smelter SO_x and visibility at Tucson and Phoenix are shown in Figure 9-19.

Table 9-3 summarizes the results of the correlation/regression analysis between yearly airport extinction (visibility) data and Arizona smelter SO_x emissions. The correlation coefficients and Student's t-statistics indicate significant statistical relationships at high confidence levels. The regression (extinction/emission) coefficients of $0.004 \pm 0.0005\ km^{-1}$/(1000 tons per day of SO_x) are remarkably consistent from site to site and represent the change in yearly median extinction associated with a given change in SO_x emissions; that is, adding 1000 tons per day of SO_x tended to increase yearly median extinction by approximately $0.004\ km^{-1}$.

Perhaps the best example of changed emission patterns is a strike that shut down the copper industry for more than 9 months in 1967-1968. In the Southwest at this time, copper production accounted for over 90 percent of the SO_x emissions, less than 1 percent of the NO_x emissions, and less than 10 percent of the conventional particulate emissions (Marians and Trijonis, 1979), and should therefore have affected visibility primarily through its contribution to sulfate loadings. Substantial decreases in sulfate occurred at five locations (Tucson, Phoenix, Maricopa County, White Pine, and Salt Lake City) within 19 to 113 km (12 to 70 miles) of copper smelters as shown in Figure 9-20. More notably, sulfates dropped by about 60 percent at Grand Canyon and Mesa Verde; these remote sites are located 325 to 500 kilometers (201 to 310 miles) from the main smelter area in southeast Arizona. Comparing measurements taken during the strike with those taken during the surrounding 4 or 6 years, Trijonis and Yuan (1978a) found a large decrease in Phoenix sulfate loadings, accompanied by a substantial improvement in visibility (Figure 9-21).

Visibility improved at almost all locations during the strike, with the largest improvements occurring near and downwind (north) of the copper smelters in southeast Arizona and near the copper smelters in Nevada and Utah. The nine locations showing statistically significant improvements are all within 242 km (150 miles) of a copper smelter. Attributing the improvement in visibility entirely to the drop in sulfate levels yields an estimated extinction efficiency of $3.9\ m^2/g$, in agreement with data in Table 9-1.

Eldred et al. (1981) studied sulfate emissions and ambient air levels before, during, and after a three-month strike beginning July 1, 1980, which shut down 9 of 11 copper smelters

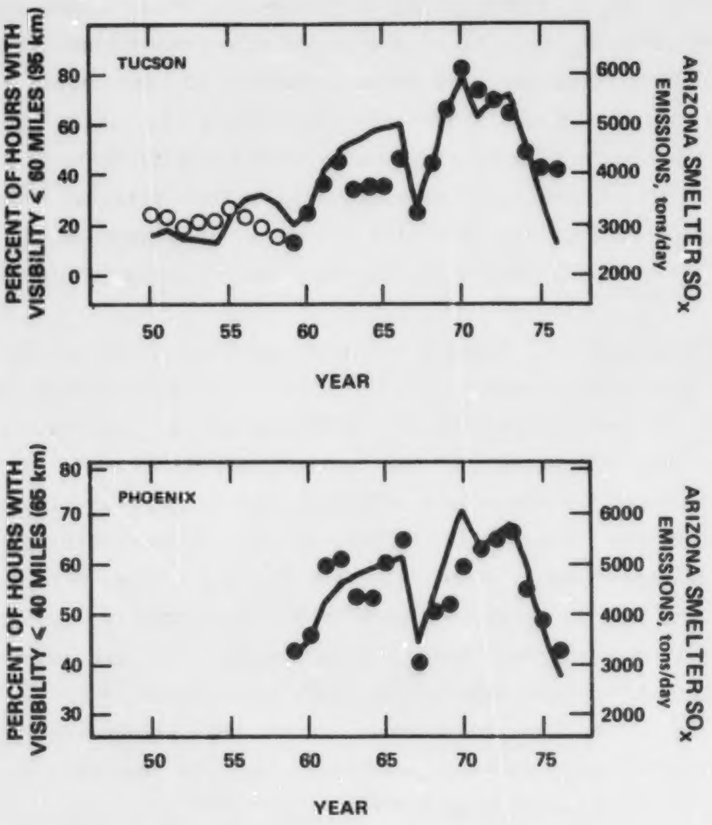


Figure 9-19. Historical trends in hours of reduced visibility at Phoenix and Tucson are compared with trends in SO_x emissions from Arizona copper smelters. Data points represent yearly percent of hours with reduced visibility. The Tucson observation site moved in 1958; although this move did not produce a statistically significant change in reported visibilities, open dots are used to distinguish data prior to 1958. Lines represent yearly statewide SO_x emissions from Arizona copper smelters.

Source: Marians and Trijonis (1979).

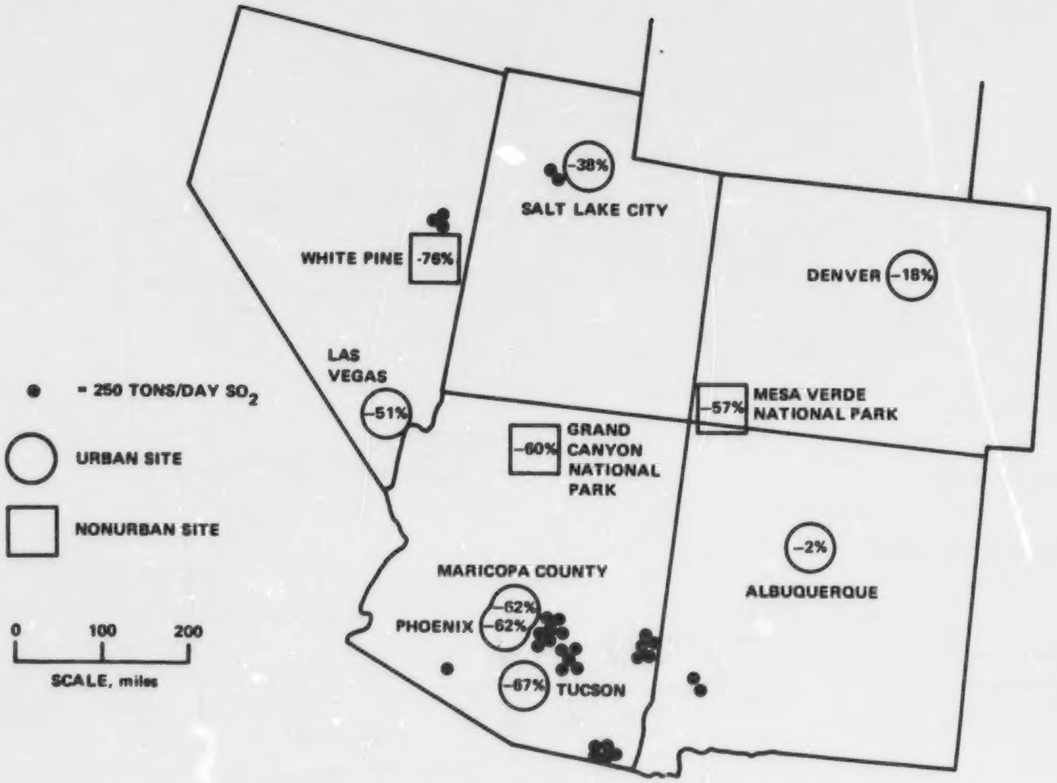


Figure 9-20. Seasonally adjusted changes in sulfate during the copper strike are compared with the geographical distribution of smelter SO_x emissions.

Source: Trijonis and Yuan (1978a).

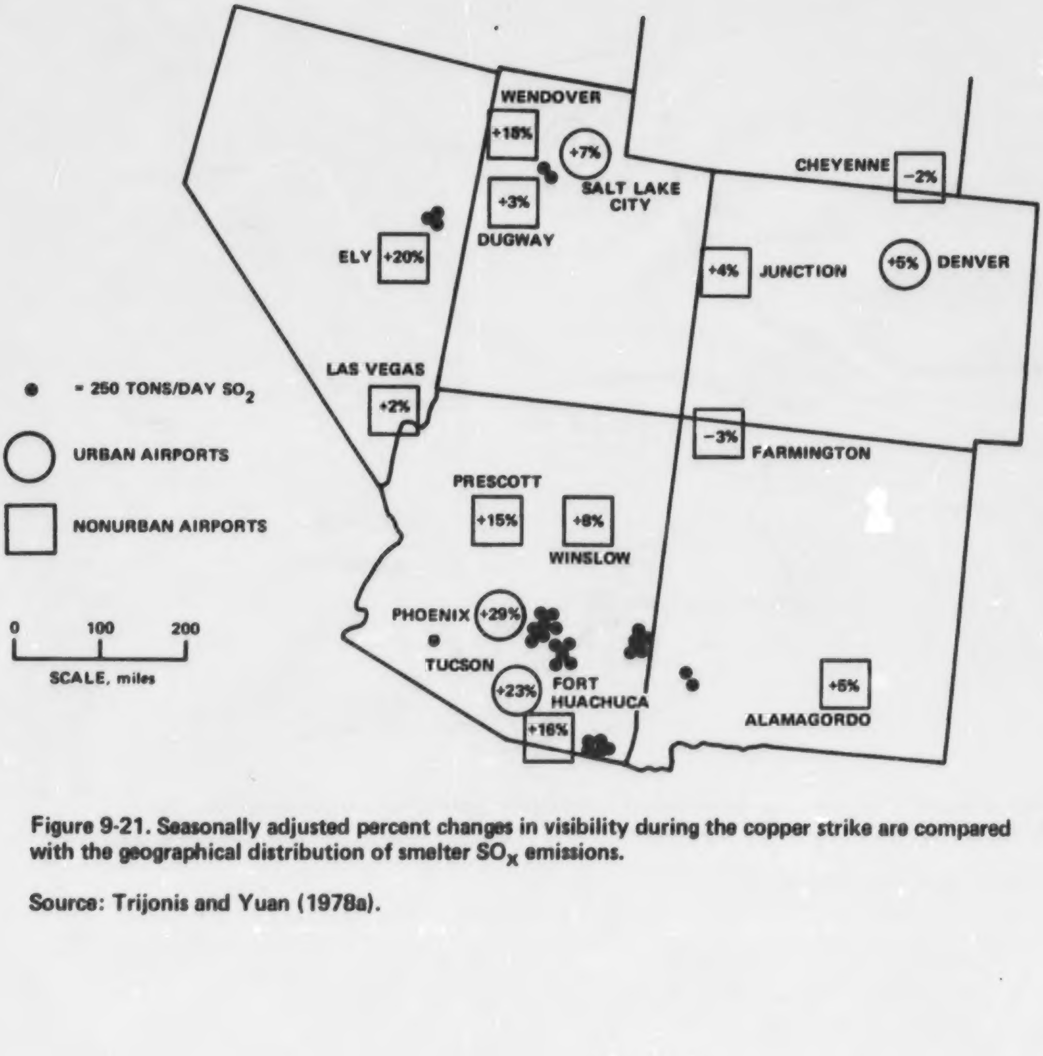


Figure 9-21. Seasonally adjusted percent changes in visibility during the copper strike are compared with the geographical distribution of smelter SO_X emissions.

Source: Trijonis and Yuan (1978a).

TABLE 9-3. CORRELATION/REGRESSION ANALYSIS BETWEEN AIRPORT EXTINCTION AND COPPER SMELTER SO_x EMISSIONS

Data set	Correlation coefficient	Regression coefficient extinction/emissions, $(10^4\ m)^{-1}/(1000\ TPD^a)$	t-statistic (t ≅ 1.7 for 95% confidence) (t ≅ 2.5 for 99% confidence)
Tucson (1950-75)	0.91	0.035	11.1
Tucson (1959-75)	0.88	0.038	7.2
Phoenix (1959-75)	0.81	0.041	5.4
Winslow (1948-73)	0.68	0.047	4.5
Prescott (1948-75)	0.70	0.031	5.0
Prescott (1948-69)	0.70	0.039	4.4

[a] TPD = tons per day

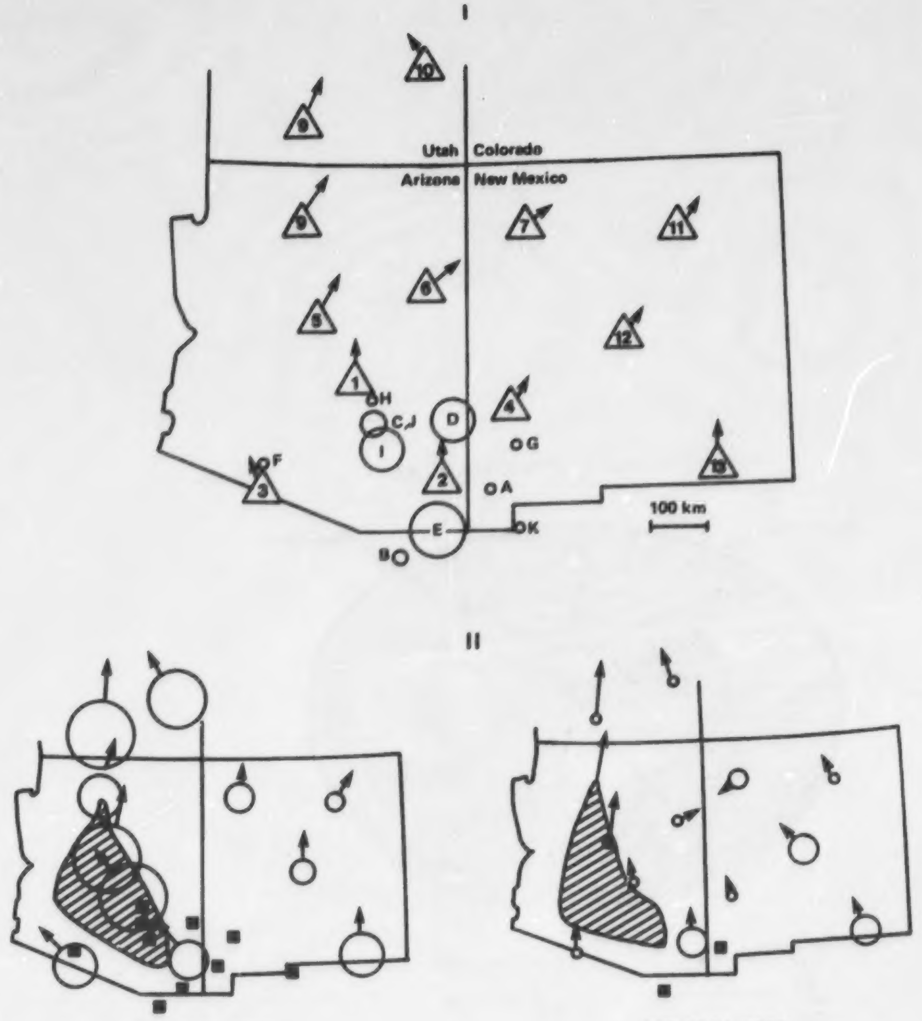
Source: Marians and Trijonis (1979)

near the southern borders of Arizona and New Mexico. At each sampling site, 72-hour concentrations were measured for fine and coarse PM, (i.e. those less or greater than 2.5 μm). The location of samplers and smelters and mean surface wind vectors are given in Figure 9-22. The investigators concluded that:

1. The largest sulfate concentrations at remote sites in Arizona and southern Utah from August 1979 to June 1980 were accompanied by wind trajectories from copper smelters;
2. During periods of southerly winds, the smelter SO_2 emissions produced significant increase in sulfate levels throughout northern Arizona and southern Utah. Comparisons of one period prior to the strike with a similar one during the strike indicated that the smelters increased sulfate in the region from 0.5 $\mu g/m^3$ to 5 $\mu g/m^3$.
3. The sulfate impact from the smelters was increased in summer by higher conversion rates of SO_2 to sulfate;
4. During the copper strike of the summer of 1980, mean sulfate concentrations throughout Arizona decreased 50 percent to 90 percent from levels of the previous summer.

Macias et al. (1981b) studied the contributions of major source types to air quality and visibility in the desert Southwest United States. The study focused in the Utah-Arizona border near the Grand Canyon and Canyonlands National Parks. In their emission inventory estimates, southern Arizona copper smelters were the largest source of S emitted into the atmosphere of this region, while southern California was the major source of NO_x and gaseous hydrocarbons. Using measurement data from VISTTA and other studies (Macias et al., 1980) and the CAPITA Monte Carlo model (which treats pollutants as quantized masses with horizontal advection within a single well-mixed vertical layer), Macias and coworkers determined an impact of southern California smog on the air quality of the Southwest United States. They also suggested that during two documented summer incursions southern California was the source of secondary pollutants, (i.e., sulfate and organic aerosols and ozone were formed primarily over southern California and not during transport across the desert). Both incursions were accompanied by a shift toward larger sulfate particle sizes. The authors emphasized that sulfates and other secondary aerosols were responsible for much of the regional haziness in the study area (Figure 9-23). They argued that if the bulk of sulfates was formed over southern California and not during transport across the desert, then a ton of sulfur emitted in south California would have a greater impact on visibility throughout the Southwest than would a ton of sulfur emitted in the desert.

Cahill et al. (1981) compared ambient air PM in Arizona and Utah with simultaneous telephotometer and nephelometer measurements of standard visual range. Using data from the VISTTA program during June 26 to July 11, 1979, the authors examined the relationships between



SEP 24-27, 1979

SEP 12-14, 1980

Figure 9-22. I: The locations of sampling sites (△) and smelters (○) and the mean surface wind vectors (↑) at each sampling site from August 1979 through September 1980; Smelter B at center of circle has area proportional to estimated 1975 SO_2 emissions. Smelters A and B did not stop production. The restart dates for the others are: 9/16 (C), 10/9 (D,E,F,G), 10/25 (H), 10/31 (I), 12/1 (J), 1/6 (K). The sampling sites are 1 (Tonto), 2 (Fort Bowie NHS), 3 (Organ Pipe Cactus), 4 (Gila Cliff Dwelling), 5 (Montezuma Castle), 6 (Petrified Forest), 7 (Chaco Canyon), 8 (Grand Canyon), 9 (Bryce), 10 (Canyonlands), 11 (Fort Union), 12 (Gran Quivera), 13 (Carlsbad Caverns). Sites 1-3,5-8, and 12-13 began in early August 1979, while the others began in late September. The mean wind vectors at each sampling site are for the period from August 1979, to September 1980. II: Sulfate concentrations and wind data for two comparison sample periods. The area of each circle is proportional to the sulfate concentration, with the maximum area representing $6 \mu g/m^3$. The arrows represent the 72-hour mean wind vectors at each site. The shaded area is the envelope of 48-hour wind trajectories ending at Grand Canyon. The locations of operating smelters are denoted by solid squares.

Source: Adapted from Eldred et al. (1981).

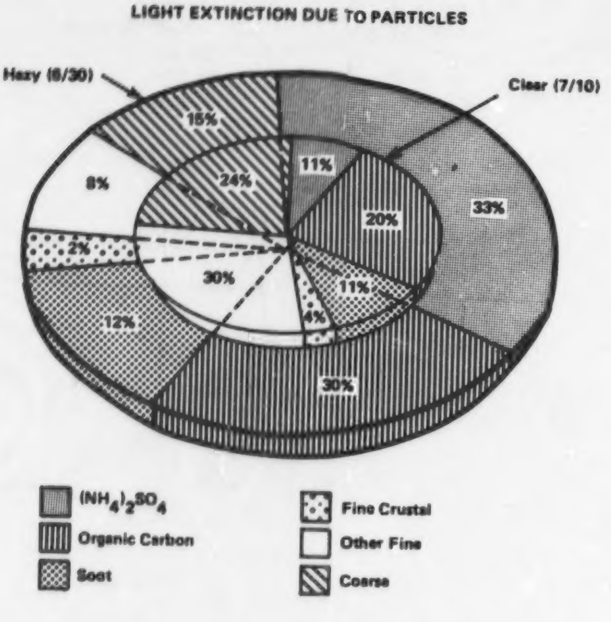


Figure 9-23. Particle light extinction ($\sigma_{sp} + \sigma_{ap}$) budget for the low visibility southern California incursion (June 30) and a clear day (July 10). On the hazy day ammonium sulfate and organic carbon accounted for 63% of the particle light extinction, while on the clear day these species accounted for only 31%.

Source: Macias et al. (1981 a and b).

particle size range and chemical composition and visual ranges. They noted that episodes of reduced visibility were associated with increased sulfur levels or increased levels of fine soils in the 0.5 to 3.5 μm range.

Altshuller (1973) has noted an increase over the past decade in sulfate concentrations at nonurban sites in the Eastern United States, which is not inconsistent with the decreasing trend in nonurban site median visibilities noted by Trijonis and Yuan (1978b). Unfortunately, the historical record of sulfate concentrations extends back only to the mid-1960's. Within the Eastern United States, over 90 percent of the SO_x emissions are associated with the combustion of coal and oil. One apparent conclusion is that visibility reduction is currently due in large part to increases in sulfate aerosols, which are formed primarily from coal combustion-related SO_2 emissions. Examination of the trends and changing spatial distributions of coal use should be comparable with the change in the light-extinction coefficient.

Air pollutants emitted over the Eastern United States result mainly from the combustion of fossil fuels: coal, oil products, and gas. The great spatial and seasonal variability of haziness (inverse of visibility) prompted Husar et al. (1979) to examine the patterns of coal consumption in the Eastern United States over the past few decades. For comparison with coal consumption estimates, visibility data are expressed in terms of a light extinction coefficient, σ_{ext}, via the Koschmieder formula: $\sigma_{ext} = 3.9/V$.

Figure 9-24 illustrates the striking similarity between summertime average haziness and coal use within the Eastern United States over the past three decades. As shown in Figure 9-25, in 1951 the haziness was most pronounced in the winter, when the coal consumption was highest. By 1974, there was a shift toward a summer peak, coincident with the increasing summer use of coal. Such coincident behavior alone cannot establish cause-effect relationships. Nevertheless, it is instructive to examine the more detailed spatial and temporal patterns of coal use and haziness (specifically, extinction).

Since 1940, the trend in coal consumption has been more pronounced in the summer than in the winter (Figure 9-26; U.S. Bureau of Mines, 1933-74); since 1960, summer coal use has grown by about 5.8 percent per year compared with 2.8 percent per year for winter coal demand. Monthly coal combustion peaked in the winter in the early 1950's, but the seasonal pattern had shifted to a summer peak by 1974 (Figure 9-25). The corresponding regional trends of haziness in the Eastern United States (Figure 9-27; Husar et al., 1979) exhibit changes similar to those of coal combustion.

In the Ohio River Valley region, the winter (quarter 1) average extinction (σ_{ext}, km^{-1}) decreased slightly, whereas the spring (quarter 2) average increased. The summer (quarter 3) extinction increased from roughly 0.25 in the 1950's (a visibility of 10 miles) to about 0.4 in the 1970's (a visibility of less than 6 miles). Fall (quarter 4) extinction remained essentially unchanged. The summer average in New England increased from about 0.2 to 0.3,

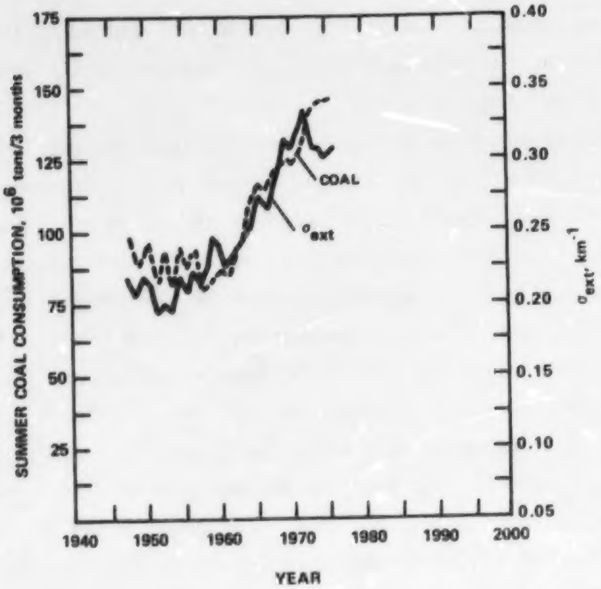


Figure 9-24. Compared here are summer trends of U.S. coal consumption and Eastern United States extinction coefficient.

Source: Adapted from Husar and Patterson (1980).

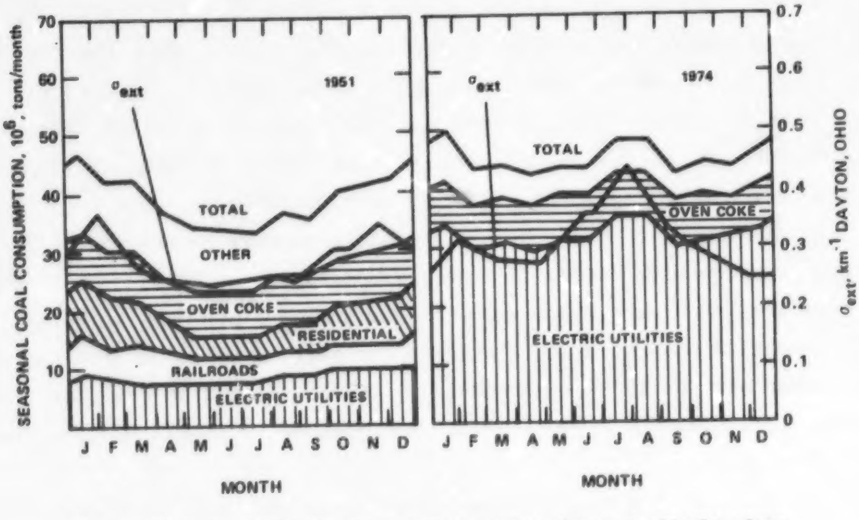


Figure 9-25. In the 1950's the seasonal coal consumption peaked in the winter primarily because of increased residential and railroad use (left figure). By 1974, the seasonal pattern of coal usage was determined by the winter and summer peak of utility coal usage (right figure). The shift away from a winter peak toward a summer peak in coal consumption is consistent with a shift in extinction coefficient from a winter peak to a summer peak in Dayton, OH, for 1948—52.

Source: U.S. Bureau of Mines, Minerals Yearbooks 1933—1974.

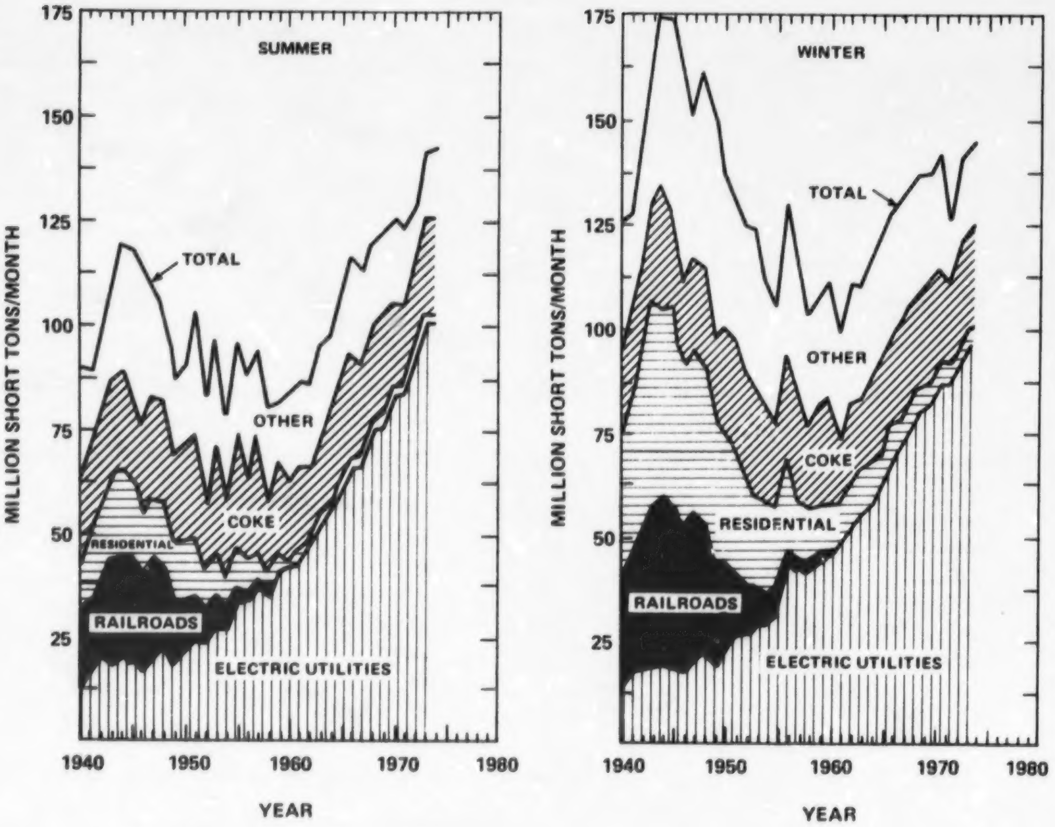


Figure 9-26. In 1974, the U.S. winter coal consumption was well below, while the summer consumption was above, the 1943 peak. Since 1960 the average growth rate of summer consumption was 5.8 percent per year, while the winter consumption increased at only 2.8 percent per year.

Source: U.S. Bureau of Mines, Minerals Yearbooks 1933-1974.

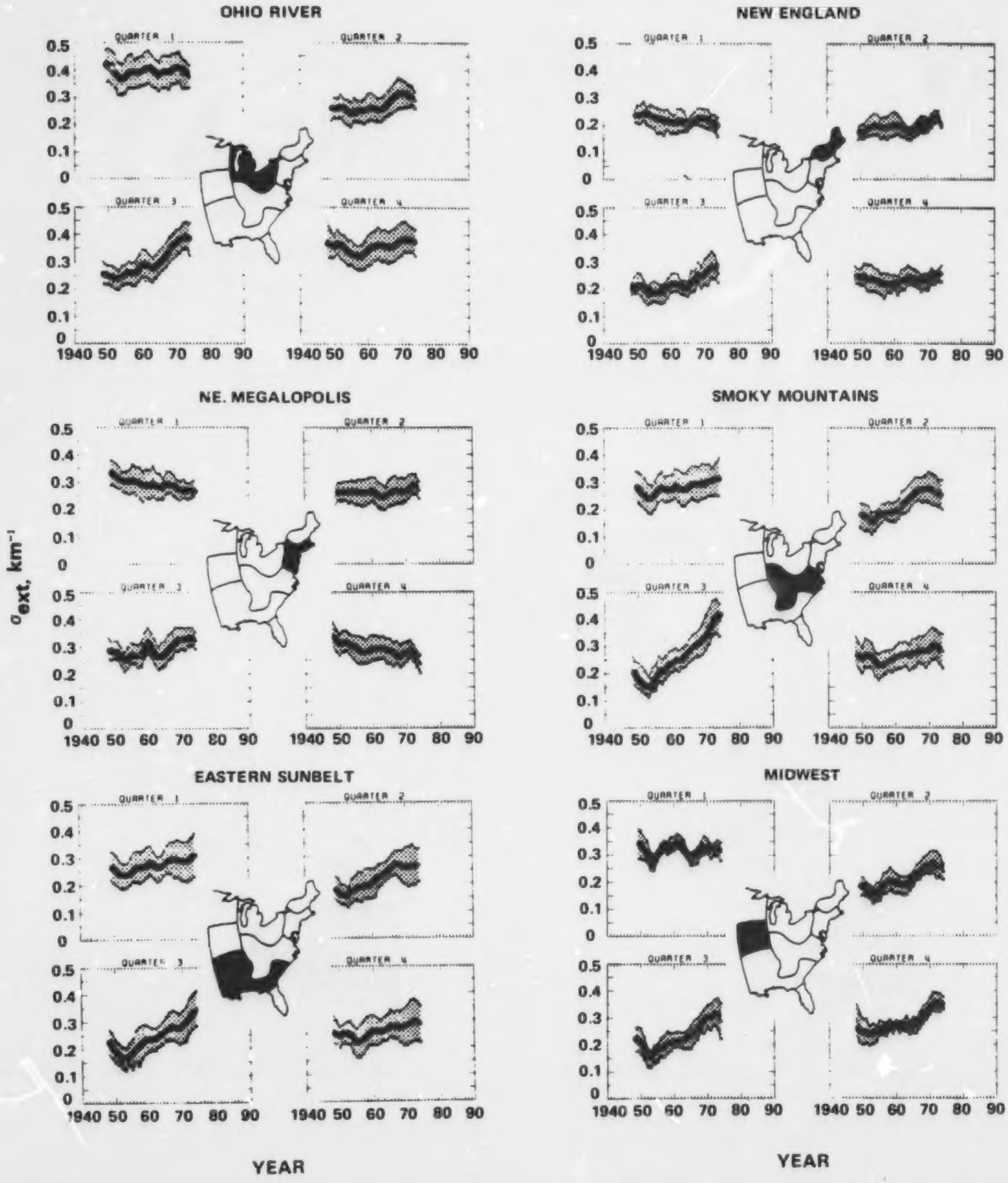


Figure 9-27. Trends in the light extinction coefficient (σ_{ext}) in the Eastern United States are shown by region and by quarters; 1 (winter), 2 (spring), 3 (summer), 4 (fall).

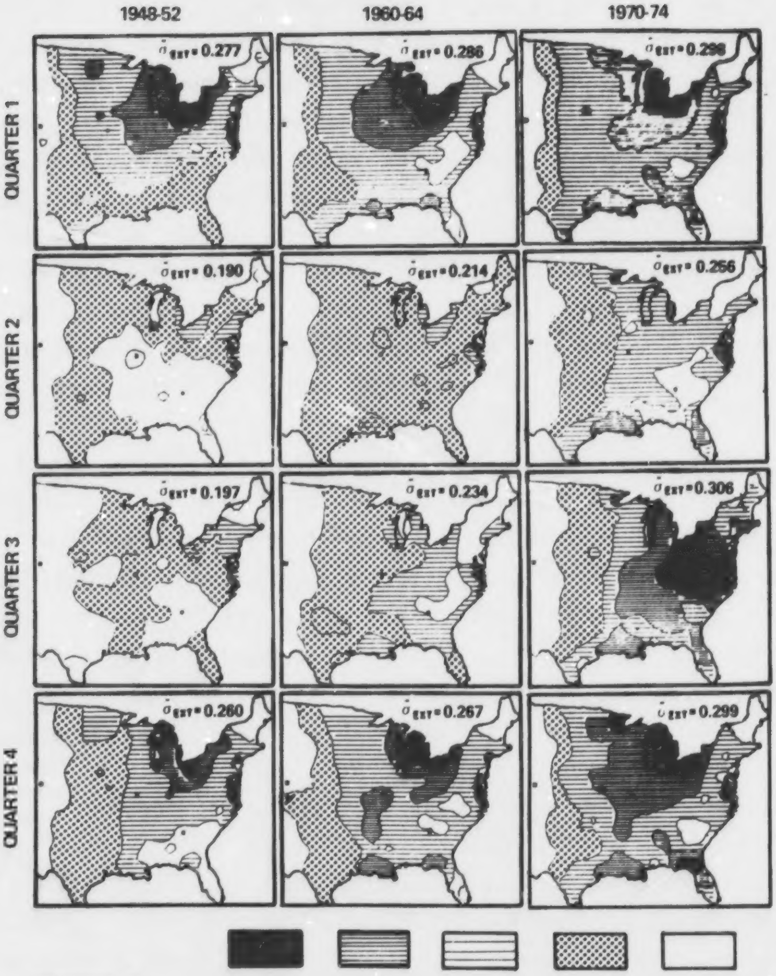
Source: Husar et al. (1979).

corresponding to a reduction in visibility from 20 to 13 kilometers (12 to 8 miles). The Northeast megalopolis region shows a general decline in haziness during quarters 1 and 4, whereas quarters 2 and 3 display a slight increase from σ_{ext} = 0.27 (15 kilometers, 9 miles) to 0.3 (13 kilometers, 8 miles) x km^{-1}. The Smoky Mountain region displays a strong increase in the average summer quarter extinction coefficient from about 0.16 to 0.4, corresponding to visibility deterioration from 15 to 6 miles (24 km to 10 km). Smaller but still pronounced increases are noted for quarters 2 and 4. Evidently the Smoky Mountains have become appreciably "smokier" over the past 20 summers. The eastern Sunbelt region has an increased haziness for all quarters, most pronounced being the summer quarter, with an increased extinction from 0.2 (12 miles; 19 km) to 0.35 (7 miles; 11 km). In the Midwest, extinction during the first quarter fluctuated slightly, with no discernible trend. The spring and fall quarters have increased appreciably, but summer values have nearly doubled, from 0.15 to 0.3 (16 to 8 miles; 26 to 13 km). The spatial shifts of Eastern United States haziness are displayed in greater detail in Figure 9-28 (Husar et al., 1979).

Husar and Holloway (1981) and Sloane (1980) have used historical data to determine trends in visibility in the Northeastern and Mideastern United States. Husar and Holloway (1981) analyzed visibility observations at Blue Hill, Massachusetts, reported from 1889 to 1958. The observations focused on the discernibility of three distant mountains. In their analyses, one of the issues they hoped to resolve was whether visibility was ever as good anywhere in the Eastern United States as current visibility in the Southwest, owing to the northeast's higher humidities, greater vegetation densities conducive to secondary aerosol formation, and hygroscopic marine aerosols. They noted that two peaks of haziness corresponded roughly with two peaks of combined coal and wood burning (1910-1920 and 1940-1950). The variability of the data, however, precluded a conclusion about the relationship between the noted increase in haziness and the burning of fossil fuels.

Sloane (1980) investigated yearly and seasonal patterns of visibility in the mideastern United States from the 1948 to 1978. Cumulative percentile and ridit analyses of the data showed that high-growth urban areas had declining visibility levels, while large metropolitan, low-growth areas, and the Appalachian region had fluctuating but improved visibility. For high-growth areas, significant declines occurred during the spring and summer quarter. Although an overall improvement in visibility was shown generally for the Mideastern United States during the summer quarter, the visibility was still below pre-1960 levels.

Wolff et al. (1981) studied three haze episodes associated with maritime tropical air traveling northward from the Gulf of Mexico and reported that all three originated in the Northeast and Midwest United States. The episodes were identified by traces of O_3 and by the temporal changes in light-scattering effects of particles, primarily sulfates. Air quality data measured by a mobile laboratory and by satellite photography and wind trajectories were used to determine movements of the haze.



	■	▤	☰	▩	□
EXTINCTION COEFFICIENT, km^{-1}	>0.36	0.3-0.36	0.24-0.30	0.18-0.24	<0.18
VISIBILITY, miles	<6.6	6.6-8	8-10	10-13.3	>13.3
(km)	(< 11)	(11-13)	(13-16)	(16-21)	(>21)

Figure 9-28. The spatial distribution of 5-year average extinction coefficients shows the substantial increases of third-quarter extinction coefficients in the Carolinas, Ohio River Valley, and Tennessee-Kentucky area. In the summers of 1948-1952, a 1000-km size multistate region around Atlanta, GA, had visibility greater than 24 km (15 miles); visibility had declined to less than 13 km (8 miles) by the 1970s. The spatial trend of winter (first quarter) visibility shows improvements in the Northeast megalopolis region and some worsening in the Sunbelt region. Both spring and fall quarters exhibit moderate but detectable increases over the entire Eastern United States.

Source: Husar et al. (1979).

During the first and second episodes, August 5 to 9 and 14 to 19, 1980, respectively, transport patterns were nearly identical. Hazy areas in the Midwest and Northeast followed the 850 mbar (about 1570 meters) streamlines eastward and then southward in a clockwise direction to the Gulf coast. This movement was followed by a return toward the southern Midwest from the flow of the surface high-pressure system above the 850 mbar level. During the third episode, the hazy air mass travelled counterclockwise to the Gulf Coast, apparently induced by Hurricane David. Subsequently, it was directed clockwise around a high-pressure system that moved down from Canada into the Eastern United States and was transported back to the Midwest.

9.4.1 Natural Versus Manmade Causes

Vision in the natural, unpolluted atmosphere is restricted by blue sky scattering, by curvature of the earth's surface, and by suspended liquid or solid natural aerosols. Important sources of natural aerosols include water (fog, rain, snow), windblown dust, forest fires, volcanoes, sea spray, vegetative emissions, and decomposition processes. The particle-free atmosphere scatters light and limits visual range to about 320 kilometers (200 miles) at sea level.

Dark objects, such as distant mountains, when viewed in daytime through a particle-free atmosphere, appear bluish because blue light is scattered preferentially into the line of sight. Bright snow-covered mountain tops or clouds on the horizon can appear yellow to pink because the atmosphere scatters more of the blue light from bright "targets" out of the line of sight leaving the longer wavelength colors. The actual visual range in the particle-free atmosphere is also limited by the earth's curvature. Thus, Rayleigh scattering is seldom the limiting factor in the detection of the most distant objects (i.e., the visual range). Rayleigh scattering is, however, important in reduction of visual texture and in bluish coloration of distant dark visual targets. Moreover, air scattering is solely responsible for the blue color of the nonhorizon sky.

A recent study in the Shenandoah National Park (Ferman et al., 1981) estimated that 14 to 22 percent of σ_{ext} was attributable to natural causes: sulfates and associated water 3 to 11 percent; organics, 5 percent; Rayleigh scattering, 5 percent; and crustal dust, 1 percent. Thus, 78-86 percent of σ_{ext} was estimated to be of anthropogenic origin. The sulfate contribution estimate was based on other studies in supposedly remote locations, while the organic and crustal contribution estimates were inferred from actual Shenandoah measurements. The 11-percent sulfate estimate was based on measurements in remote South Dakota; because of general sulfate contamination of the northern hemisphere (Lawson and Winchester, 1979), this value may be more of a background value than due to natural sources. The 3-percent figure (from measurements in South America) is probably more realistic. The natural organic contribution was estimated by assuming all elemental carbon to be anthropogenic and the anthropogenic organic/elemental carbon emission ratio to be 1.5. Uncertainties in this ratio and carbon speciation analyses may have depressed the 5-percent estimate. In the worst case, assuming all measured carbon was organic and from natural sources, the contribution would rise to 10 percent. The sum of these slightly revised percentages (including a more realistic value of 3 percent for Rayleigh scattering for visual observations) indicates that 12 to 17 percent of σ_{ext} was attributable to natural causes. Using Koschmieder's formula with K = 3.9, visual ranges of 60 to 80 kilometers in the absence of anthropogenic influence are estimated.

Fog is a natural phenomenon that can reduce the visual range to nearly zero. It is characterized by high liquid water content, typically over 1000 $\mu g/m^3$, dispersed in droplets with a mean diameter of several μm or more. In "natural" fogs all colors are scattered and absorbed about equally, so the atmosphere appears white (Husar et al., 1979).

The historical frequency of fogs in the Continental United States reveals considerable geographic variability (Figure 9-29). Windward coastal areas experience the highest frequency. Most inland portions of the United States west of the Appalachians can expect fewer than 20 days of fog per year, with less than 5 days of fog annually in the arid West.

With the exception of coastal and mountainous regions, fogs are rare during the summer months. Fogs tend to be localized events lasting a few hours at most, commonly during the early morning hours. On an hourly basis, fogs exist less than 1 percent of the time (Conway, 1963). Thus, the overall contribution of fog to the degradation of visual air quality is small, and it is an insignificant cause of reduced visibility during the daylight hours.

Thunderstorms, other rainfall, and snow can also reduce visibility. East of Nevada, most of the United States experiences from 30 to 50 days of thunderstorm activity each year. Such storms are most common on summer afternoons. Since thunderstorms are usually intense but brief, they also contribute to visibility reduction less than 1 percent of the time on an annual basis.

Snow is a major natural impediment to visibility. It is an important factor in many regions of the North and in some mountainous areas, where blowing snow occurs from 1 to 12 percent of winter hours (Conway, 1963). During the winter months, snowstorms may account for most of the hours of reduced visibility, and certainly may dominate the episodes of extremely low visibility in winter months.

The natural contribution of fog, thunderstorms, snow, and other forms of precipitation can thus cause severe degradation of visual air quality. With few exceptions, however, these intense but infrequent events do not dominate the average visual range within the Continental United States; typically, only a small percentage of the hours involve storms or fog.

In the arid West, the contribution of windblown dust to degradation of visual air quality is an important problem. Because human activities that disturb natural soil surfaces add significantly to windblown dust, dust storms are only partially natural phenomena.

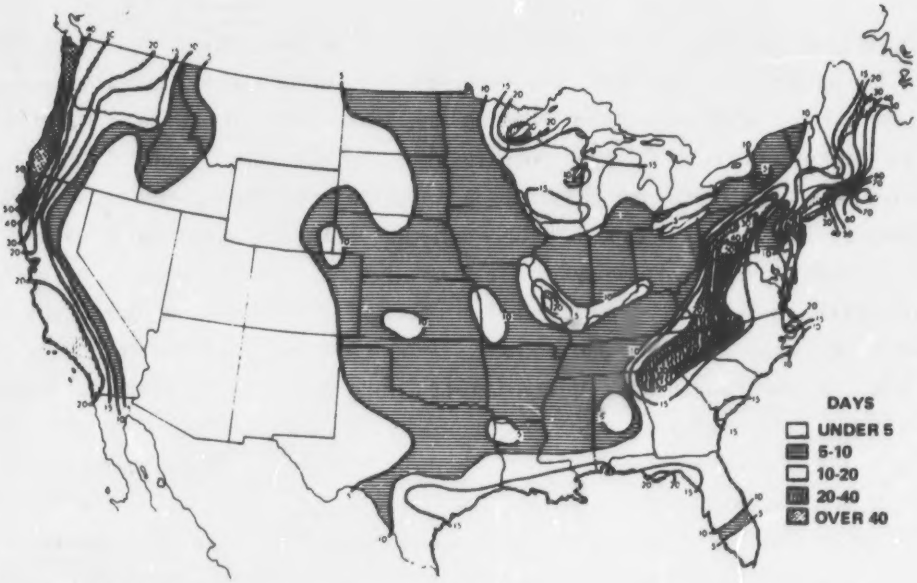


Figure 9-29. Average annual number of days with occurrence of dense fog. Coastal and mountainous regions are most susceptible to fog.

Source: Conway (1963).

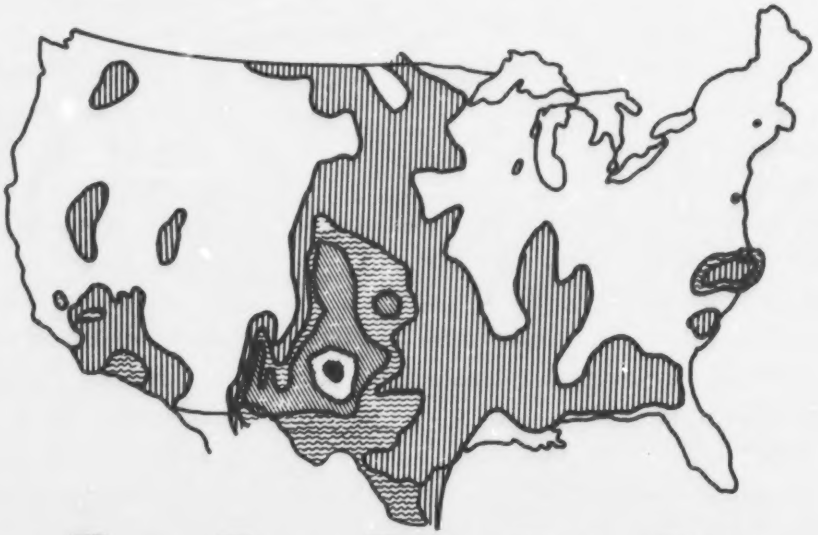


Figure 9-30. Annual percent frequency of occurrence of wind-blown dust when prevailing visibility was 7 miles or less, 1940-1970. Dust is a visibility problem in the Southern Great Plains and Western desert regions.

Source: Adapted from Orgill and Sehmel (1976).

The suspension of particles from the surface is determined by cohesiveness of the particles to the underlying material, the force of the surface wind, and the topography of the surface layer. The ideal situation leading to suspension of surface material is a dry, crumbling, or disturbed crust in flat terrain without vegetation. Agitation of such surfaces by strong winds and turbulence can transform a pristine arid atmosphere into a dust storm with severely reduced visibility. Suspended crustal material in a dust storm usually consists of coarse solid particles with volume mean diameters of tens of micrometers or more. Patterson and Gillette (1977) found that the optically important fugitive dust particles include those up to 40 μm in diameter. Orgill and Sehmel (1976) have analyzed the frequency of occurrence of dust storms in the Continental United States in great detail, based on National Weather Service observations of windblown dust and sand associated with visibility of 7 miles or less. The peak hours for dust are noon to 8 p.m., during the period of maximum thermal turbulence. Forested, coastal, and mountainous regions have few, if any, episodes. The Pacific coast has high (>0.1 percent) incidence of dust only in the San Joaquin Valley and the Los Angeles Basin. Western desert areas in eastern Washington, western Nevada, Utah, New Mexico, and Arizona are also prone to dust. The highest dust frequency is in the southern Great Plains, where windblown dust is a serious problem up to 3 percent of the time (Figure 9-30).

9.5 THE EVALUATION OF IMPAIRED VISIBILITY

In the previous sections, fine particles have been established as a primary cause of visibility degradation. Further, manmade emissions are the greatest contributors to levels of fine particles. In this section, evaluation of visibility as an important component of public welfare will be discussed. The evidence clearly shows that poor visibility lessens the quality of life, that it presents a hazard to transportation, and that people are willing to pay to improve it.

Although the value of visibility may be intangible, it can be measured to some extent. From the lack of consistent results for measuring aesthetic qualities such as visibility, values can be categorized according to psychological, social/political, or economic criteria. Psychological criteria relate to individual need and benefit associated with visibility. The U.S. Environmental Protection Agency (1979) reported that certain psychological benefits, real or perceived, are associated with a person's wish to preserve the option for a clear view of a scenic area and with the knowledge that certain pristine areas exist, regardless of any intent to visit those areas. The psychological benefits have been little studied and will not be discussed further.

Social and political criteria for evaluating visibility relate to community opinions and attitudes about visibility. These have typically been examined by surveys or questionnaires to determine whether members of a community hold visibility to be a public good (a nonexclusive asset, consumption of which by one individual does not preclude consumption by another). The third category of criteria for evaluation is economic. Both surveys of willingness to pay for good or improved visibility and studies of property value differentials as a measure of willingness to pay have been employed to determine what monetary value should be assigned to good or improved visibility. The latter two criteria for evaluation have been used in several studies and are discussed later in this section.

In addition to the aesthetic properties of good or improved visibility, inconveniences or hazards to ground and air transportation have been associated with reduced visibility caused in part by fine PM. These effects have both social and economic aspects and are also discussed.

9.5.1 Social Awareness and Aesthetic Considerations

Reduced visibility is an aesthetic effect that may occur at pollution levels below those that cause measurable health and other welfare effects. Although these aesthetic effects are difficult to measure, several studies have shown that the public considers local air pollution a nuisance and is willing to pay to reduce it. Although the importance of visibility relative to other air pollution effects has not generally been examined in most of these studies, both general expectations and some studies suggest that visibility has a major influence on the social awareness of air pollution.

According to Barker (1976), for example, the proportion of the public bothered by air pollution increases as the level of PM increases. Unlike most gaseous pollutants, which can be perceived only at high concentrations, PM causes a variety of physical stimuli at lower levels, such as direct smoke emission and visibility reductions, and therefore is a sensitive indication of pollution.

The 1969 Air Quality Criteria Document (AQCD) for PM recognized the importance of public concern over pollution by PM. Although research in this area was limited at that time, it still pointed up a heightened social awareness about air pollution and a willingness "to act to abate the nuisance." The 1969 AQCD for PM singled out a study in St. Louis, Missouri (U.S. Department of Health, Education, and Welfare, 1965; Schusky, 1966; Williams and Bunyard, 1966), to serve as an example of the association between public opinion and air pollution. Within the approximate range of 50 to 200 $\mu g/m^3$ PM (measured as TSP), an expression was derived to relate the percentage of the St. Louis population concerned about air pollution (y) and the annual geometric mean of PM levels (x)

$$y \cong 0.3x - 14 \qquad (9\text{-}17)$$

Accordingly, at levels of 80, 120, and 160 $\mu g/m^3$ TSP, about 10, 20, and 33 percent, respectively, of the public were bothered by the pollution. The same studies indicated that the public became aware of pollution before it is regarded as a nuisance, with 30, 50, and 75 percent indicating awareness at the same PM levels noted. Booz, Allen and Hamilton (1970) also confirmed this awareness by noting that there was a higher proportion of residents in high TSP-level areas compared with residents of low pollution areas who believed their neighborhoods were dirty.

The degree of concern by the public varies according to several factors: the nature of the pollutants, the extent of exposure, and perceiver characteristics (e.g., age, education, occupation). Wall (1973) examined public perception of air pollution in three British communities. These communities were similar in that they were coalfield localities, they were sites of significant air pollution problems caused by domestic coal use, and were the focus of a national effort to improve air quality. Findings were based on respondents' definition of air pollution and air pollutants. Virtually all those included in the sample offered definitions of air pollution and pollutants that indicated their awareness of the problem. Wall noted that health effects of air pollution were rarely included in the definitions, although other, more visible, effects were mentioned. Wall concluded that people were much more aware of particulate pollution than gaseous pollution.

Wall also attempted to determine what behavioral adjustments people would make when faced with a threatening air pollution episode. Although many respondents indicated they favored "direct" action, (e.g., staying indoors or closing windows), 26.7 percent were unsure of what a person could do or felt they could do nothing. One behavioral choice was complaint to a third party. Although this was a common response, Wall noted that the actual ratio of "complaint potential to complaint performance" was low. He explained that "many people do not know who to approach and have doubts about the concern and efficiency of those in authority."

Flachsbart and Phillips (1980) offered a comprehensive analysis of human response to air quality. One result of this empirical investigation was the development of an observer-based air quality index (OBAQI), which incorporated three air quality measures demonstrated to relate most significantly to reported observations of air quality. Visibility was one of three highly significant parameters in Los Angeles County, the site of the study.

In testing the effectiveness of this perception-based index, the author found that people accept a certain level of air pollution before defining it as "smoggy," the sixth value of a seven-value scale ranging from extremely clean to extremely smoggy. They also found that residents of areas with relatively clean air may be more sensitive to changes in air quality than people accustomed to pollution.

9.5.2 Economic Considerations

The value of the "wilderness experience" and the importance of preserving the natural heritage have long been recognized in the United States (U.S. Environmental Protection Agency, 1979). Long-range visibility, particularly in areas like the Southwest with its natural vistas, may be what economists call a public good: an asset that is non-exclusive (U.S. Environmental Protection Agency, 1979; Fox et al., 1979). Economists have made some progress toward quantifying the values of visibility using dollars as a measure. According to Rowe and Chestnut (1981) several economic methods are being applied and developed to estimate the value of improvements in visual quality from air pollution control. These techniques, referred to as visibility benefit analyses, can be used to estimate the dollar value of changes in visibility. Analyzing the economic effects of improved visibility requires an understanding of the benefits and their monetary measures. The concepts of visibility as an economic good and of monetary benefits from visibility improvements may be classified through application of consumer demand theory. Consumer demand theory asserts that an individual derives well-being, or what economists call utility, from the consumption of goods and services and that any change affecting utility, such as improved visibility at a favorite park, has a value. The monetary measure of benefits for an improvement in visibility indicates how much the increase in utility is worth. Analyzing how individuals react to changes in visual quality may reveal the value they place on these changes. A visibility benefit analysis attempts to determine what individuals would be willing to pay for a change in visual quality if it were possible to purchase it.

Two approaches have been used to measure the value of visibility: (1) asking people to establish values directly through the use of hypothetical, or contingent, market situations; and (2) using actual market and air quality data to determine the relationship between price and visibility. The usual version of the contingent market approach is the bidding method, in which participants are asked to indicate their maximum willingness to pay (WTP) or minimum willingness to accept compensation (WTA) to obtain or prevent a change in air quality. In the market approach to measuring the value of visibility, studies are based on the supposition that if air quality varies across an area and if people are willing to pay more for a residence with better air quality, the amount they are willing to pay can be inferred from the price differences between properties when all other influences except air quality are accounted for.

The results of the dozen or so visibility benefit analyses that have been conducted, including both contingent and actual market approaches, show that the value people place on good or improved visibility is substantial. Although the circumstances of each study differ, the results are broadly consistent, providing evidence that estimation procedures are valid (Rowe and Chestnut, 1981).

Using the first approach, Brookshire (1979) explains the iterative bidding technique as "a direct determination of economic values from data which represent responses of individuals of contingencies posited to them via a survey instrument." The iterative bidding technique in its current form was first developed and applied by Randall et al. (1974) in the Four Corners region of the Southwest. Three contingencies were considered: (1) limited visibility reductions and a view of a power plant with limited visible emissions; (2) moderate emissions from the plant, moderate visibility reductions, and moderate existence of unreclaimed spoil banks and transmission lines; and (3) extensive emissions, visibility reductions, and unreclaimed spoil banks and transmission lines. Unfortunately, this selection of scenarios prohibits disaggregation of results into component values for visibility, power plant location, and unreclaimed spoil banks and transmission lines. A mean reduction in sales tax of $85 per household was required to make bidders accept scenario 3 instead of scenario 1; and reduction of $50 was required to accept 3 in place of 2 (U.S. Environmental Protection Agency, 1979). Biases are associated with these techniques; however, no bias tests were conducted in this experiment. Because bids were requested without actual payment, there may have been incentives to misrepresent one's bids, or the questions may have distorted responses.

Three other studies employing the iterative bidding technique in the Southwest are reviewed here: the Lake Powell study, by Brookshire et al. (1976); the Farmington study, by Rowe et al. (1980a,b); and the South Coast Air Basin study, by Brookshire et al. (1979; 1980).

The Lake Powell study (Brookshire et al., 1976), conducted in 1975, considered the visual impact of large power plants and visible smoke plumes in view from a predominantly recreational site. Recreationists and a few local residents were asked what they would be willing to pay to prevent construction of an additional plant if only the plant would be visible and if both the plant and pollution would be visible. Average bids by users of the recreation area, in terms of additional user fees per day, were between $0.87 and $2.11 to prevent an additional plant and between $1.75 and $3.38 to prevent an additional plant and pollution. The lowest average bids were made by residents, while the highest average bids were made by remote campers. Aggregate bids indicate that the benefits of preventing visibility degradation at that site alone (there are also several other recreation areas in the vicinity) were $400,000 to $700,000 yearly.

The Farmington study (Rowe et al., 1980a,b), conducted in 1977, was concerned with the impact of coal-fired electric generation in the Southwest. The focus of this study was the perception of reduced visibility by local residents in and near Farmington, New Mexico. Three visibility levels were illustrated by photographs of long-distance landscape views from Farmington showing (1) visual range of about 120 kilometers (75 miles), somewhat better than current conditions, (2) visual range of about 80 kilometers (50 miles); and (3) visual range of about 40 kilometers (25 miles). The average monthly bid by residents to prevent deterioration in visual range from 120 kilometers (75 miles) to 80 kilometers (50 miles) was $4.75, and the bid to prevent deterioration from 120 kilometers to 40 kilometers (25 miles) was $6.50. Nonresident recreationists in the area were willing to pay an average of $3.00 and $4.00 in additional user fees per day for the same scenarios. Aggregate benefit estimates for the study area over 35 years assuming a 10 percent discount rate, were $14.2 million for preventing deterioration of condition 1 to 2 and $19.2 million for preventing deterioration condition of 1 to 3. One of the purposes of this study was to test for biases in the bidding process. Hypothetical, starting-point, and information biases were detected. Additionally, it was found that, contradictory to theoretical expectations, WTA bids were much larger than WTP bids. This outcome was attributed to differences in implied property rights in the two questionnaire procedures used. Rowe et al. (1980a,b) also reviewed a number of iterative bidding and property value differential studies as they might be applied to visibility values.

The South Coast Air Basin study (Brookshire et al., 1979, 1980), conducted in 1978, was an application of the bidding method and the property value approach to the same urban areas to compare the results of these two techniques. The scenarios used in the bidding method were illustrated with photographs of two views in Los Angeles showing (1) poor air quality as a visual range of about 2 miles, typical in much of the area; (2) fair air quality as a visual range of about 12 miles, the predominant condition in the area; and (3) good air quality as a visual range of about 28 miles. Residents in 12 different communities, each categorized as having poor, fair, or good air quality, were interviewed. Average monthly bids in communities with poor air quality were between $11 and $22 to obtain fair conditions. In communities with fair air quality, average monthly bids to obtain good conditions were between $5 and $28. In communities with good air quality, residents offered average bids of $18 to $67 per month to obtain good air quality in the entire region. The mean average household bid was approximately $30 per month for a 30-percent improvement in air quality. For all proposed changes, aesthetic, acute health, and chronic health components each constituted about one third of the bid. Brookshire et al. (1979) also reported the difference in property values based on aesthetic consideration afforded by air pollution in six pairs of neighborhoods in the South Coast Air Basin of Southern California. In this study, the WTP based on property value differentials was about $40 per month per household for a 30-percent improvement in air quality. This amount is comparable with the iterative bidding results of $30 per month. For both survey and property value studies, the estimates ranged from $20 to $150 per month per household. Additionally, 22 to 55 percent of the aggregate bids for all areas were for aesthetic effects, the major component of visibility evaluation noted also by Flachsbart and Phillips (1980) for the same region.

According to Rowe et al. (1980a,b), while much progress has been made in visibility benefits valuation, much more work is needed. Early studies did not establish the link between reduced visibility and actual physical parameters of visibility. In addition, other variables such as health or other welfare effects also confound the valuation of visibility benefits. (Some of these effects are discussed in Chapter 10.) Certain inherent variables may confound estimating the value of visibility based solely on site value. For example, the perception of visually dirty air may trigger an association with unhealthy air, thereby confounding aesthetic effects with health effects. Pollutants other than PM may also confound the benefits analyses. Finally, studies of the economic value of visibility have only begun to address the urban situation.

9.5.3 Transportation Operations

This section discusses effects on transportation safety and convenience. Both automobile and aircraft safety and operations may be affected by pollution-related reductions in visual range. Highway engineers use a concept called Recommended Sight Distances (RSD) to determine visual range requirements for safe driving, passing, hill grading, etc. At recommended speeds, these distances provide adequate time for driver reaction and stopping. At 70 miles per hour (113 km/hr) the RSD is about 400 yards (365 m). At 55 miles per hour (88 km/hr), it is under 300 yards (274 m). Even with very high levels of humidity, which can greatly intensify the effect of pollution upon visual range (U.S. Environmental Protection Agency, 1979), very high pollutant concentrations are required to reduce visual range to less than one-fourth mile (440 yards; 400 m). For example, concentrations exceeding 1150 $\mu g/m^3$ of fine mode particles (< 2.5 μm) would be required to reduce visual range to less than 440 yards (400 m) at 90 percent RH for a highly scattering and absorbing aerosol, as depicted in Figure 9-18.

Improved visibility may result in air travel that is safer and involves fewer delays. Air traffic is controlled using either Instrument Flight Rules (IFR) or Visual Flight Rules (VFR). Generally when visual range (roughly measured as the average over three directions) falls to below 3 (4.8 km) miles and there is less than a 1000-foot (300 m) ceiling, air travel is judged unsafe in controlled airspace using VFR, and only IFR-rated pilots and planes may fly. Commercial aircraft continue to operate, but certain general aviation flights are grounded. Between 70 to 80 percent of general aviation aircraft are equipped to fly IFR (Federal Aviation Administration, 1981a), but only 35 percent of certified general aviation pilots are instrument rated (Federal Aviation Administration, 1981b). Out of 357,500 total certified general aviation private airplane pilots, 36,500 (10 percent) are instrument rated (Federal Aviation Administration, 1981b). General aviation (fixed-wing aircraft) logged approximately 40 million hours of flight time in 1979 distributed according to the following use (in millions of hours): personal ~9.2; business ~8.7; instructional ~6.4; executive ~4.7; rental ~4.1; air taxi ~3.6; aerial application ~2.1; industrial ~0.8; and other ~0.8 (Federal Aviation Administration, 1980b). Personal and rental amounted to 13.3 million hours, or 33 percent of the total general aviation hours. Since only 10 percent of the private pilots are instrument rated, it follows that about 30 percent of general aviation flights operating in controlled airspace would likely be grounded during visibilities less than 3 miles (4.8 km).

A U.S. Senate staff report (U.S. Congress, 1963) noted that air pollution was both hazardous and costly to aircraft operations. Recognizing that fog slows air traffic, the author noted that the addition of pollutants to the fog often reduced visibility enough to ground aircraft not equipped with blindflying instruments and to delay traffic at busy airports. Additionally, the report cited a review by the Civil Aeronautics Board of 1960 record cards, representing one-third of all U.S. aircraft accidents in 1962, and attributed six accidents to obstruction of vision by smoke, haze, sand, or dust. Two of the six planes were classified as large, and one was a commercial carrier. The report noted that if the sample study was representative, 15 to 20 plane crashes in 1962 could have been linked to poor visibility. The staff report stated:

> Pollution contributes to fog formation and undoubtedly aggravated visibility problems in which natural conditions play a major role. Where accidents are attributed to impaired visibility due to weather, it is likely that in many instances the risk had been increased by air pollution.

A 1967 report on air pollution abatement activity concluded that "interstate air pollution in the New York - New Jersey metropolitan area results in visibility restrictions endangering the safety of persons in interstate travel, both by land and by air, and it causes inconvenience and economic loss to the public and to transportation companies, due to disruption of traffic schedules" (National Center for Air Pollution Control, 1967).

The Federal Aviation Administration (1978) in the Airmen's Information Manual notes that failure to see and avoid obstructions during flight is one of the 10 factors most frequently associated with aircraft accidents. The manual notes that caution should be exercised when visibilities are within the 3 to 4 mile (4.8 to 6.4 km) limit. The National Transportation Safety Board (National Transportation and Safety Board, 1978b) also ranked weather as a major factor related to total United States certified air carrier accidents from 1969 to 1978. In 1978, briefs of 928 total and 322 fatal accidents involving weather as a cause/factor list low ceilings and visibilities as probable causes and factors associated with the fatal accidents the majority of times (National Transportation and Safety Board, 1978b).

The Air Traffic Service of the Federal Aviation Administration has developed a performance measurement system for many major terminals (Federal Aviation Administration, 1980a). In this system standards were set for individual airports according to runway configuration and visibility levels. Under IFR and VFR conditions, the number of operations per hour (arrivals and departures) are tabulated according to runway configuration. At most of the major airports covered in the report, significant reductions in operations per hour occurred during IFR conditions, i.e., when visual range is below 3 miles (4.8 km). In addition some airports may require nonvisual approaches at visibilities between 3 and 5 miles, depending on the minimum vectoring altitude (about 1000 to 1500 ft [300 to 460 m] ceiling). In this case, the arrival and possibly the departure operations may be affected marginally at visibilities greater than 3 miles (4.8 km).

To determine how often visibility is reduced to below 3 miles at airports throughout the country, information from the National Weather Service historical data base was compiled and plotted for 147 sites in the United States. Because only half mile increments are reported for the 2- to 3-mile (3.2 m to 4.8 km) range, 3-mile (4.8 km) or less range is reported here. The data consisted of local standard time midday readings (1200, 1300, or 1400 hours) for six 5-year periods: 1951-1955, 1956-1960, 1961-1965, 1966-1970, 1971-1975, and 1976-1980. Additionally, 3-month quarters were reported separately within the 5-year periods. Table 9-4 shows the frequency of poor midday visibility (i.e., 3 miles [4.8 km] or less) in the northeastern, southeastern, or midwestern regions of the United States. Except for the West Coast and Texas, the frequency of poor midday visibilities of 3 miles (4.8 km) or less in most areas

TABLE 9-4. SEASONAL AVERAGE PERCENT OF TIME WHEN MIDDAY VISIBILITY WAS 3 MILES (4.8 km) OR LESS AT U.S. AIRPORTS FROM 1951 to 1980[a]

Location	Quarter 1 (Jan.-Mar.)		Quarter 2 (Apr.-June)		Quarter 3 (July-Sept.)		Quarter 4 (Oct.-Nov.)	
	all weather	good weather[b]	all weather	good weather[b]	all weather	good weather[b]	all weather	good weather[b]
Northeast	16.2	2.4	7.6	2.1	8.5	4.8	12.0	2.3
Southeast	12.5	1.1	5.6	1.9	7.7	4.7	9.4	1.3
Midwest	19.8	4.9	6.7	2.5	6.0	3.5	15.6	4.9

[a] Based on daily measurements over the entire 30 years. Data from the U.S. Weather Service.

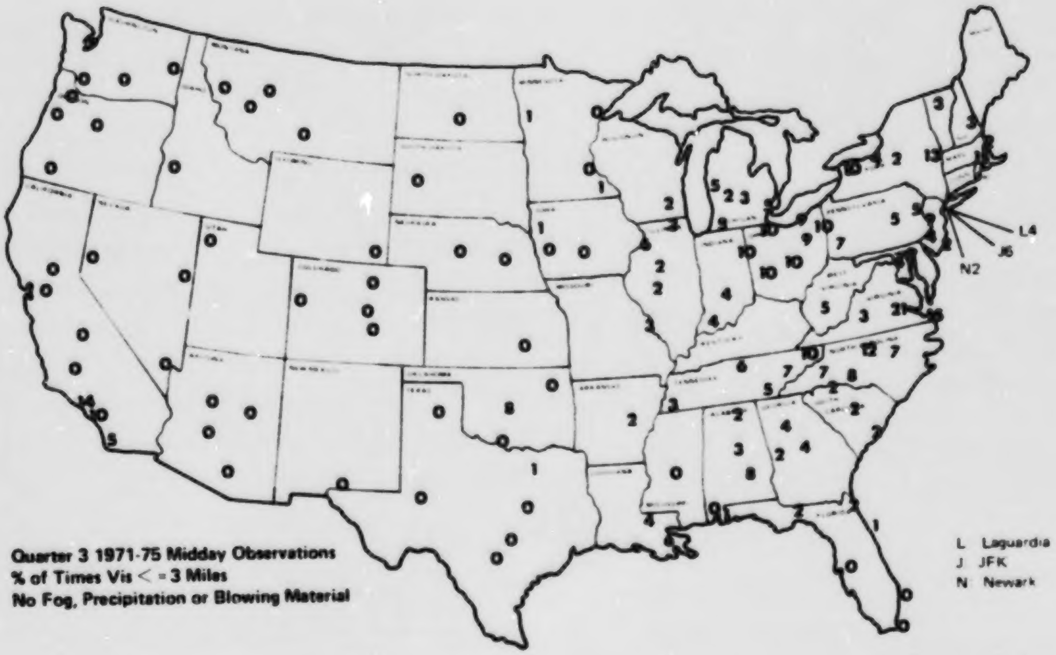
[b] Absence of fog, precipitation, or windblown material such as snow, sand, or dust.

of the Western United States is low. Data are reported for poor visibility under all weather conditions and under good weather conditions (i.e., in the absence of fog, precipitation, blowing material). Under good weather conditions, poor visibility occurred most frequently in the summer quarter, accounting for 60 percent of all instances of poor visibility in this season. At other times of the year (Quarters 1, 2, and 4), poor visibility during good weather represented only 20 percent of the total number of instances of poor visibility. Figures 9-31 and 9-32 show the location of the sites and percent of times and occurrences during quarter 3 (July-September) that visibility was 3 miles (4.8 km) or less for the two most recent 5-year periods, 1971-1975 and 1976-1980. Each quarter included approximately 460 measurements. In these figures, neither fog, precipitation, nor blowing material (e.g., sand and dust) was the cause of the incidences of reduced visibilities. For all six 5-year periods (1951-1980) Table 9-5 lists the occurrences of 3-mile (4.8 km) or less visibilities at midday for all causes and for causes attributed primarily to pollution (i.e., during periods when no fog, precipitation, or blowing material was present) for the summer quarters for 26 airport sites in the United States. In most cases, the percent of occurrence of low visibility reached a maximum from 1966 to 1975. Data from the National Weather Service also show that RH greater than 90 percent occurred rarely during midday at most locations throughout the United States. Examination of Table 9-5 leads to the conclusion that midday visibility reductions to less than 3 miles (4.8 km) can occur in summer months more than 50 percent of the time from causes other than fog, precipitation, or blowing material, and around 40 percent of the time when RH is about 75 percent. Presumably, then, these instances of reduced visibility are attributable to light-scattering or light-absorbing air pollutants (i.e., fine-mode PM).

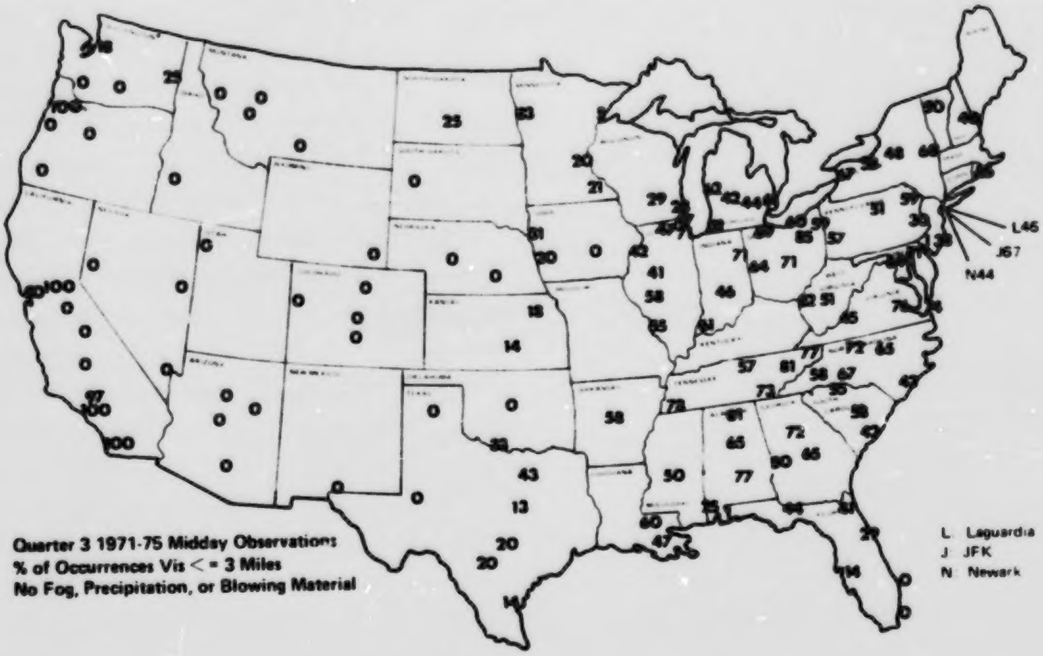
From the limited information available, it was concluded that during periods of peak air traffic fewer planes arrive and depart when visibility is below 3 miles (4.8 km) (IFR conditions) than when visibility is greater than 3 miles (4.8 km) (VFR conditions). More than half of the time that IFR conditions are in effect during the summer months, the weather is good and the RH is below 90 percent. At such times, fine-mode PM is probably responsible for the IFR conditions. The airport data listed in Table 9-5 indicate that poor midday visibility (equivalent to IFR conditions) occurred an average of 7 percent of the time during the summer quarter, from 1951-1980 in the Northeast, Southeast, and Midwest. It follows that more than half of the time that visibility was poor enough to slow midday air traffic at some airports in these regions, fine-mode PM was the cause. Thus, the overall frequency of poor airport midday visibility related to fine-mode PM was about 4 percent. By the same line of reasoning, fine-mode PM could have impeded midday air traffic only about 2 percent of the time annually.

9.6 SOLAR RADIATION

Incoming solar radiation is composed of the direct bean and the diffuse sky light arising from the light-scattering atmosphere (Figure 9-33; Gates, 1966). The relative contribution of the sky light is least at noon and greatest at sunrise and sunset. At sea level, and for a clean atmosphere, sky light contributes at least 10 percent of the total radiation.



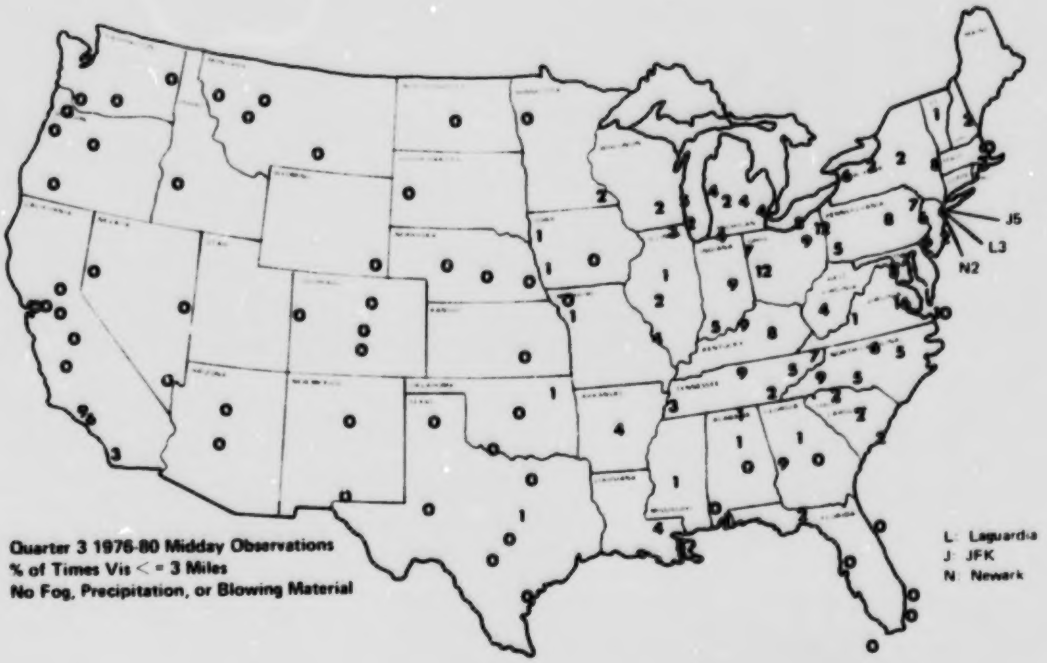
Quarter 3 1971-75 Midday Observations
% of Times Vis < = 3 Miles
No Fog, Precipitation or Blowing Material



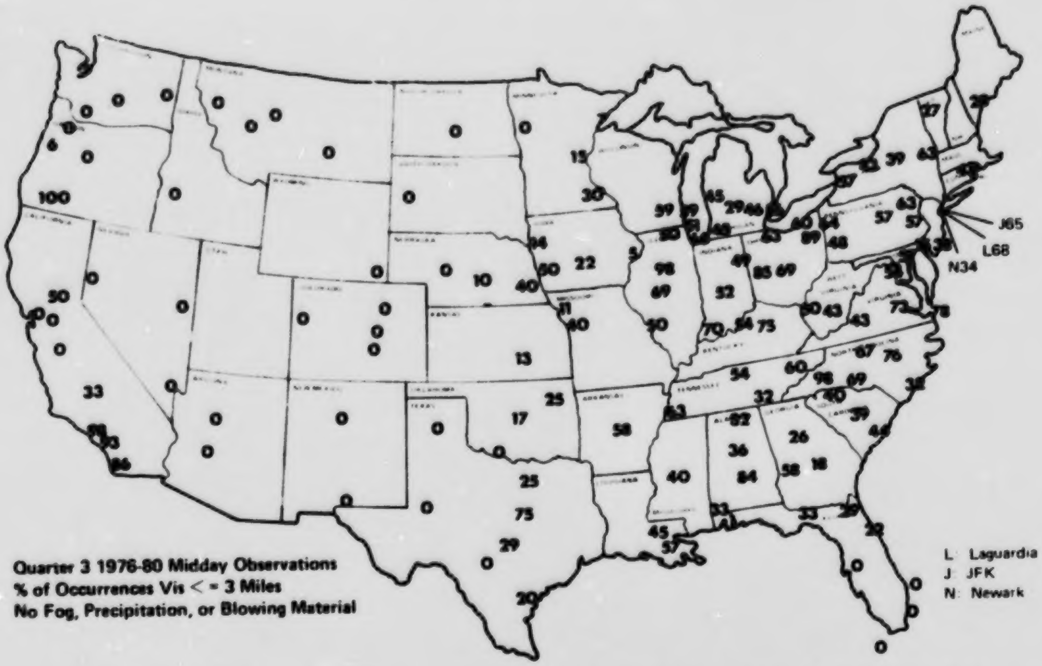
Quarter 3 1971-75 Midday Observations
% of Occurrences Vis < = 3 Miles
No Fog, Precipitation, or Blowing Material

Figure 9-31. Percent of daily midday measurements (1971-75) in which visibilities were three miles or less in the absence of fog, precipitation or blowing material.

Source: Adapted from the Historical Data Base of the National Weather Service 1981.



Quarter 3 1976-80 Midday Observations
% of Times Vis < = 3 Miles
No Fog, Precipitation, or Blowing Material



Quarter 3 1976-80 Midday Observations
% of Occurrences Vis < = 3 Miles
No Fog, Precipitation, or Blowing Material

Figure 9-32. Percent of daily midday measurements (1976-80) in which visibilities were three miles or less in the absence of fog, precipitation, or blowing material.

Source: Adapted from the Historical Data Base of the National Weather Service 1981.

TABLE 9-5. PERCENT OF VISIBILITY MEASUREMENTS AT 3 MILES (4.8 km) OR LESS AT 26 U.S. AIRPORTS DURING THE SUMMER QUARTER[a]

Location	Period	Percent of visibility measurements less than or equal to 3 miles			
		In the presence of fog, precipitation, or blowing material	In the absence of fog, precipitation, or blowing material		
		All RH's[b]	All RH's[c]	RH ≤ 75%	RH ≤ 60%
Northeast					
New York, NY (LaGuardia)	1951-1955	5.9	4.8	3.7	2.0
	1956-1960	8.3	5.2	4.3	1.5
	1961-1965	9.4	6.5	5.7	3.1
	1966-1970	14.1	10.0	8.5	4.3
	1971-1975	8.0	3.7	2.4	1.3
	1975-1980	4.1	2.8	2.2	1.1
New York, NY (JFK)	1951-1955	4.8	4.1	3.0	1.5
	1956-1960	9.1	5.4	4.1	2.0
	1961-1965	6.7	4.3	3.3	1.1
	1966-1970	12.0	8.3	6.1	1.7
	1971-1975	9.1	6.1	4.3	1.1
	1975-1980	7.4	4.8	3.5	1.3
Newark, NJ	1951-1955	6.1	3.3	2.4	0.9
	1956-1960	11.3	5.4	4.1	1.7
	1961-1965	7.6	4.1	3.3	1.1
	1966-1970	10.4	7.2	6.5	2.6
	1971-1975	5.0	2.2	1.5	0.4
	1976-1980	5.0	1.7	1.5	0.4
Pittsburgh, PA (GP)	1951-1955	3.4	1.4	1.4	0.7
	1956-1960	5.2	2.8	2.2	0.7
	1961-1965	4.3	1.5	0.9	0.2
	1966-1970	10.0	6.5	6.1	2.8
	1971-1975	12.2	7.0	6.3	4.1
	1976-1980	10.5	5.0	3.7	1.3

TABLE 9-5. (ocntinued)

Location	Period	Percent of visibility measurements less than or equal to 3 miles			
		In the presence of fog, precipitation, or blowing material	In the absence of fog, precipitation, or blowing material		
		All RH's[b]	All RH's[c]	RH ≤ 75%	RH ≤ 60%
Albany, NY	1951-1955	5.9	1.5	1.3	0.2
	1956-1960	3.9	0.4	0.2	0.0
	1961-1965	5.4	1.7	1.7	0.7
	1966-1970	14.8	10.4	9.8	5.7
	1971-1975	19.8	13.5	13.0	7.0
	1976-1980	13.7	8.7	7.2	3.9
Buffalo, NY	1951-1955	3.5	1.3	0.9	0.7
	1956-1960	4.3	1.5	1.1	0.2
	1961-1965	6.5	2.8	2.4	0.2
	1966-1970	8.7	5.4	4.6	1.7
	1971-1975	15.0	10.0	9.3	3.5
	1976-1980	10.7	6.1	4.1	2.0
Providence, RI	1951-1955	4.6	2.6	2.4	0.4
	1956-1960	8.7	2.8	1.3	0.0
	1961-1965	9.1	3.3	1.3	0.4
	1966-1970	12.4	8.9	6.7	1.5
	1971-1975	9.8	5.4	3.7	1.1
	1976-1980	5.0	2.2	1.3	0.7
Southeast					
Montgomery, AL	1951-1955	0.9	0.0	0.0	0.0
	1956-1960	1.3	0.0	0.0	0.0
	1961-1965	0.4	0.0	0.0	0.0
	1966-1970	4.3	2.2	2.0	0.9
	1971-1975	10.2	7.8	7.4	3.9
	1976-1980	9.6	8.1	7.6	5.7

TABLE 9-5. (continued)

Location	Period	Percent of visibility measurements less than or equal to 3 miles			
		In the presence of fog, precipitation, or blowing material	In the absence of fog, precipitation, or blowing material		
		All RH's[b]	All RH's[c]	RH ≤ 75%	RH ≤ 60%
Richmond, VA	1951-1955	3.9	0.9	0.4	0.0
	1956-1960	5.4	2.4	1.7	0.4
	1961-1965	7.4	2.0	1.7	0.9
	1966-1970	18.3	13.7	12.0	6.7
	1971-1975	28.0	21.3	19.6	10.7
	1976-1980	22.0	16.1	14.8	8.7
Norfolk, VA	1951-1955	2.8	0.2	0.2	0.2
	1956-1960	4.6	1.3	0.9	0.0
	1961-1965	7.4	3.7	2.4	0.2
	1966-1970	15.4	11.7	9.8	4.6
	1971-1975	20.4	15.2	13.3	3.9
	1976-1980	12.6	9.8	9.6	6.1
Greensboro, NC	1951-1955	2.4	0.4	0.2	0.0
	1956-1960	4.6	0.7	0.2	0.0
	1961-1965	4.1	2.2	0.9	0.2
	1966-1970	6.3	3.9	2.8	1.1
	1971-1975	16.5	12.0	10.0	6.1
	1976-1980	9.2	6.1	4.6	2.8
Atlanta, GA	1951-1955	0.2	0.0	0.0	0.0
	1956-1960	3.1	0.7	0.4	0.2
	1961-1965	2.2	0.4	0.4	0.0
	1966-1970	3.0	1.3	0.9	0.4
	1971-1975	5.4	3.9	3.7	1.5
	1976-1980	4.1	1.1	1.1	0.9

TABLE 9-5. (continued)

Location	Period	Percent of visibility measurements less than or equal to 3 miles			
		In the presence of fog, precipitation, or blowing material	In the absence of fog, precipitation, or blowing material		
		All RH's[b]	All RH's[c]	RH ≤ 75%	RH ≤ 60%
Bristol, TN (Tri-City)	1951-1955	0.4	0.0	0.0	0.0
	1956-1960	0.9	0.0	0.0	0.0
	1961-1965	1.3	0.2	0.0	0.0
	1966-1970	4.8	2.6	2.4	1.1
	1971-1975	12.4	9.6	8.0	2.0
	1976-1980	9.1	6.5	6.3	2.4
Memphis, TN	1951-1955	0.7	0.2	0.2	0.0
	1956-1960	1.5	0.0	0.0	0.0
	1961-1965	1.1	0.0	0.0	0.0
	1966-1970	1.3	0.7	0.2	0.0
	1971-1975	3.9	3.0	3.0	2.0
	1976-1980	5.2	3.3	3.1	2.0
Baton Rouge, LA	1951-1955	1.3	0.9	0.7	0.0
	1956-1960	2.4	0.7	0.4	0.4
	1961-1965	2.4	0.4	0.4	0.2
	1966-1970	4.3	2.4	2.2	0.9
	1971-1975	6.5	3.9	3.5	1.5
	1976-1980	8.3	3.7	3.5	1.5
Washington, DC (National)	1951-1955	2.8	1.1	0.4	0.4
	1956-1960	3.5	1.5	0.9	0.2
	1961-1965	3.5	1.1	0.9	0.0
	1966-1970	2.8	1.3	1.1	0.2
	1971-1975	4.8	2.6	2.6	1.7
	1976-1980	3.7	2.0	2.0	0.7
Lexington, KY	1951-1955	0.9	0.0	0.0	0.0
	1956-1960	2.4	0.7	0.7	0.4
	1961-1965	2.6	0.4	0.2	0.0
	1966-1970	5.7	2.0	1.7	1.3
	1971-1975	8.7	5.2	4.1	2.4
	1976-1980	12.4	9.3	0.0	3.5

TABLE 9-5. (continued)

		Percent of visibility measurements less than or equal to 3 miles			
		In the presence of fog, precipitation, or blowing material	In the absence of fog, precipitation, or blowing material		
Location	Period	All RH's[b]	All RH's[c]	RH ≤ 75%	RH ≤ 60%
Midwest					
Detroit, MI (City)	1951-1955	3.9	2.0	1.3	1.3
	1956-1960	7.8	4.8	3.7	1.5
	1961-1965	8.0	4.1	2.8	1.1
	1966-1970	12.8	9.3	8.0	5.2
	1971-1975	8.3	5.0	3.9	1.5
	1976-1980	7.4	4.1	3.7	2.2
Chicago, IL (Midway)	1951-1955	2.6	1.5	1.1	0.2
	1956-1960	5.9	4.1	3.3	1.5
	1961-1965	7.6	3.7	2.8	1.1
	1966-1970	5.7	4.1	2.8	2.0
	1971-1975	6.5	4.6	3.0	1.7
	1976-1980	8.2	5.4	3.0	2.2
Chicago, IL (O'Hare)	1956-1960	4.9	4.3	3.3	1.6
	1961-1965	5.9	2.0	1.5	0.9
	1966-1970	4.3	3.0	2.2	0.9
	1971-1975	5.9	2.8	2.0	0.9
	1976-1980	3.9	2.0	0.9	0.4
St. Louis, MO	1951-1955	2.2	0.9	0.7	0.4
	1956-1960	1.7	0.7	0.4	0.0
	1961-1965	4.3	2.0	1.3	0.7
	1966-1970	8.9	3.9	3.5	2.2
	1971-1975	6.3	3.5	2.8	1.3
	1976-1980	6.1	3.0	2.8	2.0

TABLE 9-5. (continued)

Location	Period	Percent of visibility measurements less than or equal to 3 miles			
		In the presence of fog, precipitation, or blowing material	In the absence of fog, precipitation, or blowing material		
		All RH's[b]	All RH's[c]	RH ≤ 75%	RH ≤ 60%
Southwest					
Dallas, TX	1951-1955	0.0	0.0	0.0	0.0
	1956-1960	0.0	0.0	0.0	0.0
	1961-1965	0.7	0.0	0.0	0.0
	1966-1970	1.1	0.0	0.0	0.0
	1971-1975	1.8	0.0	0.0	0.0
Dallas/Ft. Worth, TX	1951-1955	0.0	0.0	0.0	0.0
	1956-1960	0.0	0.0	0.0	0.0
	1961-1965	0.4	0.0	0.0	0.0
	1966-1970	1.3	0.0	0.0	0.0
	1971-1975	1.5	0.7	0.4	0.2
	1976-1080	0.9	0.2	0.2	0.2
West					
Denver, CO	1951-1955	0.9	0.0	0.0	0.0
	1956-1960	0.4	0.0	0.0	0.0
	1961-1965	1.3	0.0	0.0	0.0
	1966-1970	0.4	0.0	0.0	0.0
	1971-1975	0.9	0.0	0.0	0.0
	1976-1980	0.2	0.0	0.0	0.0
Los Angeles, CA (Int'l.)	1951-1955	20.9	20.7	18.9	2.0
	1956-1960	10.4	10.4	9.1	5.2
	1961-1965	17.0	17.0	15.2	0.2
	1966-1970	18.5	18.3	15.2	1.1
	1971-1975	14.1	13.7	11.7	0.0
	1976-1080	8.8	8.6	5.5	0.2

TABLE 9-5. (continued)

Location	Period	Percent of visibility measurements less than or equal to 3 miles			
		In the presence of fog, precipitation, or blowing material	In the absence of fog, precipitation, or blowing material		
		All RH's[b]	All RH's[c]	RH ≤ 75%	RH ≤ 60%
Long Beach, CA	1951-1955	21.3	20.9	20.2	14.1
	1956-1960	12.0	12.0	11.7	9.6
	1961-1965	17.4	17.2	17.2	11.1
	1966-1970	18.7	18.7	18.3	16.3
	1971-1975	10.4	10.4	10.4	7.2
	1976-1980	6.5	6.1	6.1	4.4

[a] Average number of observations during each 5-year period was 460 for the summer quarter (July, August, and September).

[b] Blowing material refers to dust, sand, or snow.

[c] Table 9-2 shows that midday RH greater than 90 percent occurs rarely; moreover, in the absence of fog or precipitation, it occurs even more rarely.

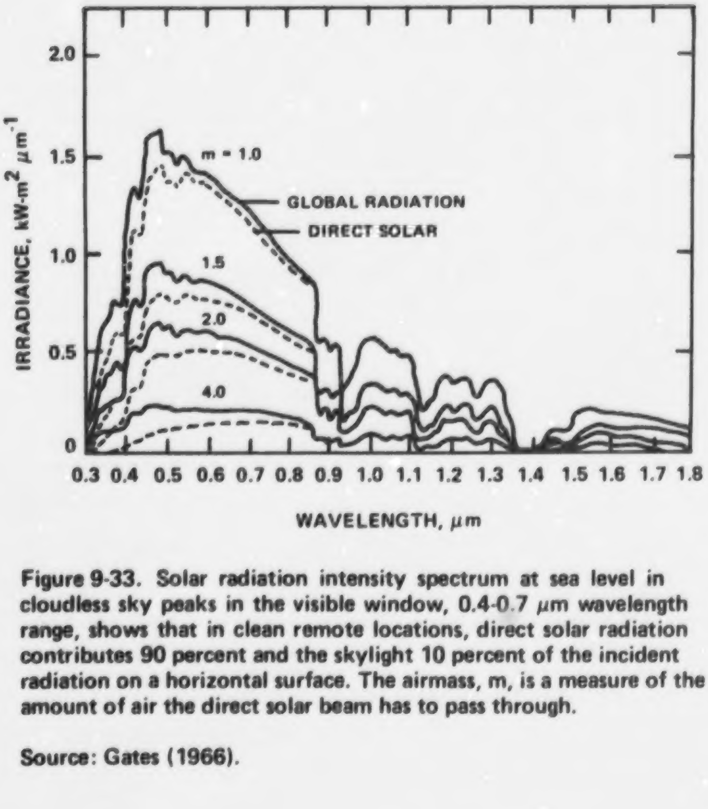


Figure 9-33. Solar radiation intensity spectrum at sea level in cloudless sky peaks in the visible window, 0.4-0.7 μm wavelength range, shows that in clean remote locations, direct solar radiation contributes 90 percent and the skylight 10 percent of the incident radiation on a horizontal surface. The airmass, m, is a measure of the amount of air the direct solar beam has to pass through.

Source: Gates (1966).

Aerosol layers in the atmosphere scatter and absorb solar radiation (Figure 9-34). Some of the scattered radiation is directed upward and lost to space; some is directed downward to the earth's surface. A fraction of the radiation may also be absorbed by aerosols, further reducing the amount of radiation reaching the surface but at the same time heating the aerosol layer itself from the earth's surface. Most of the solar radiation eventually reaches the surface, but its spectral and directional composition (that is, the "quality" of the solar radiation) may be changed by atmospheric haze (Weiss et al., 1979).

Because aerosols are not uniformly distributed in the atmosphere, their effects are spatially nonhomogeneous. First, the horizontal spatial scale encompassing aerosol source, transport, and removal in the lower troposphere is variable but often about 1000 kilometers 620 miles). The vertical spatial scale of aerosols is also quite variable, but often the particles are concentrated in a layer from 0.5 to 2 km (0.3 to 0.5 miles) meters deep at the earth's surface. Hence, the aerosol effects should be concentrated in the lowest layers of the atmosphere, especially in industrial regions.

Global-scale effects might also occur. If the effects in industrial regions are strong enough, then the heat balance of the entire earth could be influenced. Additionally, effects from long-lived aerosol, such as those in the stratosphere, might lead to direct physical effects on a global scale.

9.6.1 Spectral and Directional Quality of Solar Radiation

Figure 9-35 (McCree and Keener, 1974) shows the spectral quality of solar radiation on a clear day and on a hazy day in Texas. On the hazy day, the direct solar radiation is reduced to about one-half of that on a clear day, but most of the energy has reappeared as diffuse sky light. The net effect is that there is an overall loss of up to about 10 to 20 percent of the radiation reaching the surface.

If we take the typical backscattered fraction for regional haze aerosols to be 10 percent of scattering, and the absorption to be also about 10 percent, as suggested by the data of Weiss et al. (1979), then we can estimate the amount of energy lost from the surface, the amount lost to space, and the amount absorbed by the atmosphere. On a day with half of the direct beam transmitted, we conclude that 10 percent of the other half, or 5 percent, is lost to space, and the other 5 percent results in atmospheric heating. Together, these phenomena lead to a loss of 10 percent of the radiation. Although it is not possible to calculate accurately the influence this loss might have on surface temperature, rate of thawing of frozen ground, growing season, or other climatological measures, it is highly probable that this loss cools the ground and heats the hazy lower layers of the atmosphere. In turn, it must increase atmospheric stability, decrease convective mixing, and therefore increase the rate at which pollutants accumulate.

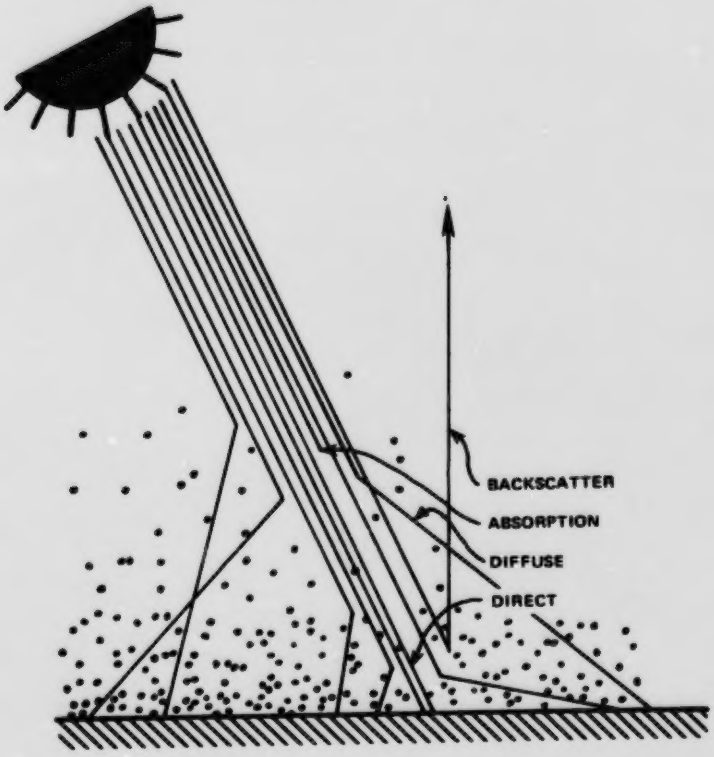


Figure 9-34. Extinction of direct solar radiation by aerosols is depicted.

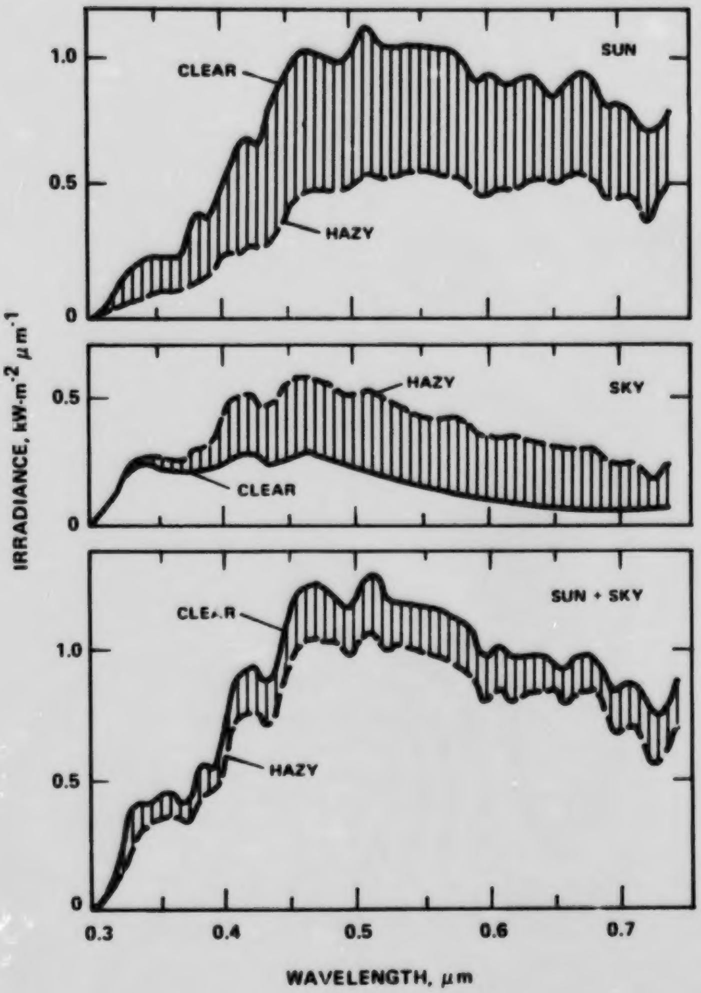


Figure 9-35. On a cloudless but hazy day in Texas, the direct solar radiation intensity was measured to be half that on a clear day, but most of the lost direct radiation has reappeared as skylight. However, there is about 20 percent of the solar radiation missing on the hazy day, some absorbed, and some backscattered to space.

Source: McCree and Keener (1974).

No detailed and routine measurements of the quality of solar radiation are available for the United States, however, the total solar energy reaching the surface is monitored routinely at many meteorological observation sites in the United States and worldwide. Unfortunately, the large variability of such data does not allow manmade aerosol effects to be distinguished from the effects of natural sources.

A data base that gives some information on the quality of solar radiation is the United States turbidity network, operated at about 40 stations in the country since 1961 (Flowers et al., 1969). If there are no clouds between the observer and the sun, the intensity of direct solar radiation for a given solar elevation depends on the variable amount of dust, haze, and water vapor in the atmosphere. The extinction produced by these constituents is called "atmospheric turbidity." Figure 9-36 (Flowers et al., 1969) shows the seasonal pattern of atmospheric turbidity in the United States at 29 sites for 1961-1966. At all 29 sites, the highest turbidity occurs in the summertime and the lowest occurs in the winter, which is consistent with the haziness pattern obtained from visibility observations (Section 9.4.2). The turbidity of the atmosphere in the United States has a strong spatial dependence. In the Southwest, with a mean annual turbidity coefficient of about 0.06, the incoming direct solar radiation is attenuated by only 13 percent, compared with Midwest values of about 20 percent. (In this and subsequent statements, only the reduction due to particles and, to a much lesser extent, by NO_2, is given.) The highest turbidity coefficients were observed in the Eastern United States where winter values of 0.1 and summer values of 0.2 were typical, meaning that about 20 to 35 percent of the direct solar beam is diverted to sky light, backscattered to space, or absorbed. Thus, by earlier assumptions, 2 to 3.5 percent is backscattered to space and another 2 to 3.5 percent is absorbed into the atmosphere.

Since the first report of Flowers et al. (1969), e.g., the turbidity data have been reported yearly by the World Meteorological Organization (1977). Comparison of the seasonal turbidity pattern for 1961-1966 and 1972-1975 is shown in Figure 9-37. Since the mid-1960's there has been a further increase of turbidity in the Eastern United States, particularly in the summer season. Currently the summer average turbidity in the region including Memphis, Tennessee, Oak Ridge, Tennessee, Greensboro, North Carolina, and Baltimore, Maryland, is about 0.3. This corresponds to a 50-percent attenuation of the direct solar beam on an average summer day. During hazy episodes, turbidity coefficients of 0.6 to 1.0 are often reported, resulting in removal of 75 to 90 percent of the solar radiation from the direct beam, with 7.5 to 9 percent lost to space, and 7.5 to 9 percent lost as atmospheric heating. One of the consequences of such hazy atmosphere is the disappearance of shadow contrasts. It is strongly suspected, but has not yet been proved, that there are effects on agricultural productivity.

The spatial distribution and trends of regional-scale turbidity in the eastern United States are consistent with the observed pattern of haziness obtained through visibility

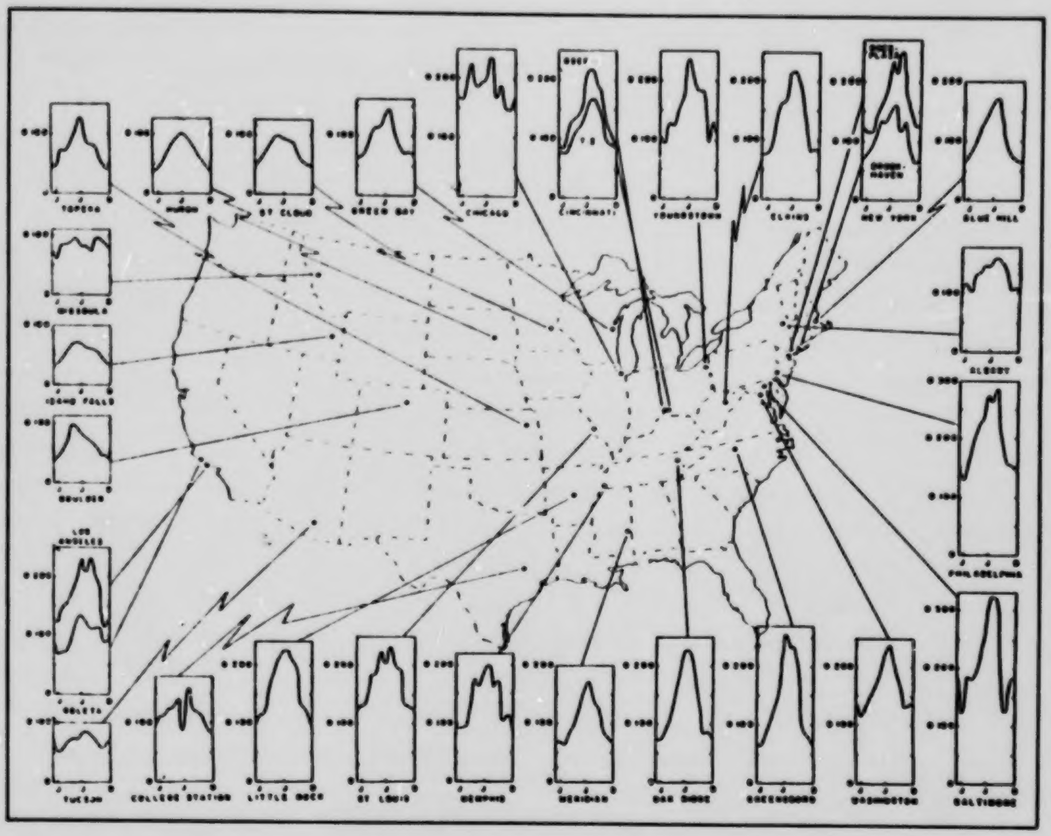


Figure 9-36. To interpret these 1961-1966 monthly average turbidity data in terms of aerosol effects on transmission of direct sunlight, use the expression $I/I_0 = 10^{-B}$, where B is turbidity and I/I_0 is the fraction transmitted.

Source: Flowers et al. (1969).

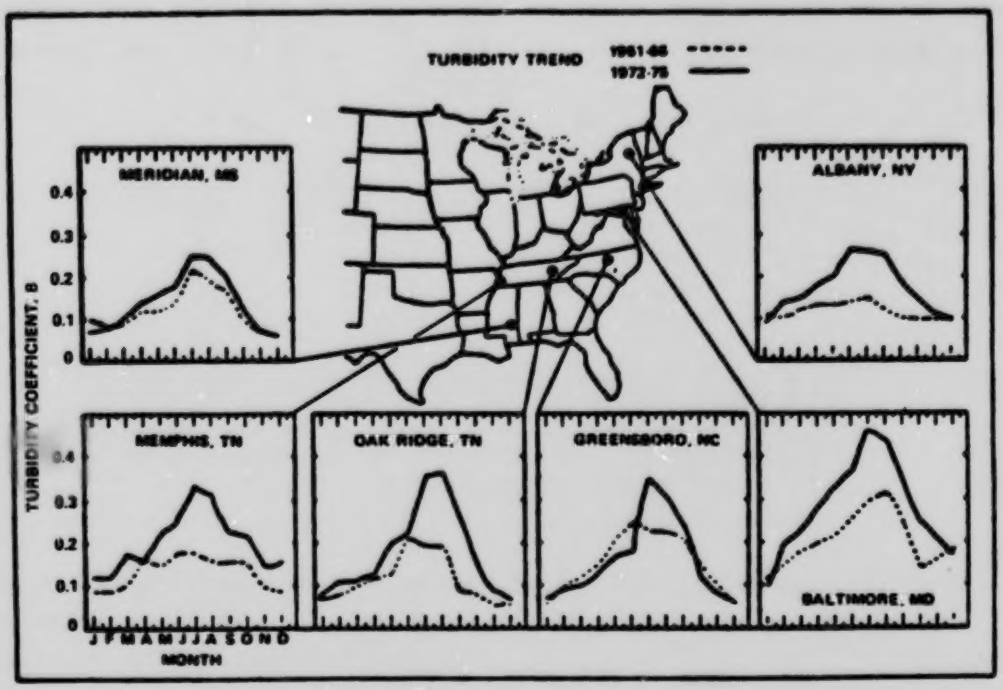


Figure 9-37. Seasonal turbidity patterns for 1961–1966 and 1972–1975 are shown for selected regions in the Eastern United States.

Source: Flowers et al. (1969), and WMO (1974 through 1977).

observations. Both the turbidity and the visibility reduction by haze in the Eastern United States can be attributed primarily to fine particles (Section 9.2.3.3). Bolin and Charlson (1976) suggest that many of these radiative effects are due to sulfates and conclude that the magnitude of effects is comparable to that summarized here.

9.6.2 Total Solar Radiation: Local to Regional Scale

Changes in the total radiant energy have been observed within urban areas. Early measurements in central city locations, primarily in Europe, showed levels typically 10 to 20 percent below surrounding rural areas. Robinson (1962) discussed some observations made in London and Vienna. In London, the deficit was considerably reduced after the implementation of a Clean Air Act. Measurements on 47 days in autumn 1973 in the Los Angeles area are summarized in Table 9-6 (Peterson and Flowers, 1977). In the St. Louis area, however, smaller urban-rural differences were observed. On 12 cloudless days in summer 1972, the average solar and U.V. fluxes at an urban site were only 3 and 8 percent, respectively, below those at a rural site about 50 kilometers from the city. The difference between the St. Louis and Los Angeles and European urban areas appears to involve both decreased urban and increased rural attenuation, and it may be that neither the city of St. Louis nor its surroundings over a wide area modify solar radiation in a manner typical of other locations.

TABLE 9-6. SOME SOLAR RADIATION MEASUREMENTS IN THE LOS ANGELES AREA[a]

Measurement	Total	UV
Minimum	4	15
Average	11	29
Maximum	20	44

[a] Values for the daily average percentage decrease of total and UV solar radiation between El Monte (urban) and Mt. Disappointment (rural).

Source: Peterson and Flowers (1977).

Angell and Korshover (1975) analyzed the solar radiation duration data (hours of sunshine) for the eastern half of the United States. Data for 1950-1970 were obtained with detectors that accumulate the time during which total illumination is above a set threshold. These are believed to be more reliable for long-term trend analysis than data from recorders of solar radiation intensity. Angell and Korshover noted some marked trends: in the southeast and south central United States, the solar radiation duration has decreased by about 4 to 6 percent; however, the north central area is increasing (Figure 9-38; Angell and Korshover, 1975). Although the authors do not attribute these trends to any specific cause,

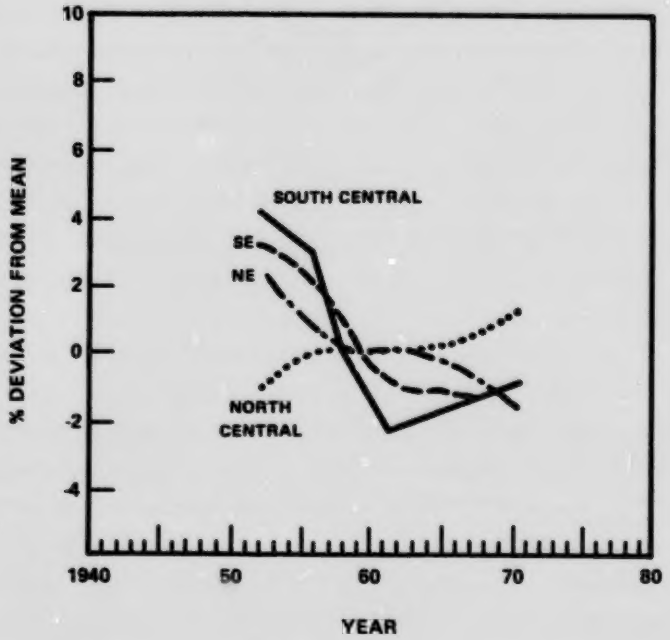


Figure 9-38. Analysis of the hours of solar radiation since the 1950's shows a decrease of summer solar radiation over the Eastern United States. There may be several causes for this trend, including an increase of cloudiness; some of the change may also be due to haze.

Source: Angell and Korshover (1975).

it should be noted that there has been an increase in haziness within that period in areas with decreased solar radiation. It is conceivable, therefore, that increased haziness causes sunshine-duration detectors to delay the turn-on time in the morning and advance the turn-off time in the evening. It should also be stressed, however, that changes in the solar radiation duration may be caused by other natural or manmade phenomena.

9.6.3 Radiative Climate: Global Scale

The attenuation of solar radiation from scattering and absorption by particles in the atmosphere is probably an important factor in climatic change. The effect could arise from redirecting the photon energy, from effects on cloud structure (perturbations of the colloidal system), or from effects on optical properties, such as increased photon retention from absorption by soot particles. A report by Hobbs et al. (1974) argued that aerosols are most likely the principal agents by which pollutants may affect weather and climate, by influencing the structure and distribution of clouds. On a global scale, they noted that the effects of manmade pollutants on climate are still a matter of debate. Others, such as Twomey (1974), have pointed up a direct connection between pollution and the number of drops in a cloud and, hence, an influence on optical thickness and reflectance of the clouds (cloud albedo); consequently, climate is affected.

The importance of PM on climate may be overshadowed by that of CO_2. A doubling of the concentration of CO_2 could result in a 1.5° to 3° C warming of the lower atmosphere, according to various predictions reviewed by the U.S. Department of Energy (U.S. Department of Energy, 1978). One set of calculations frequently cited in the scientific literature predicts a 2° to 3° C rise in the average temperature of the lower atmosphere in the middle latitudes with the doubling of the CO_2 content of the air and a three- to fourfold greater temperature increase in the polar regions (Manabe and Wetherald, 1975). Current model estimates suggest that the earth should have experienced a few tenths of a degree of warming since the late 1880s due to the increase of CO_2 concentrations from about 290 to the current 335 ppm. However, it appears either that natural variations currently are large enough to mask the expected CO_2 effect on temperature (U.S. Department of Energy, 1980; and Stuiver, 1980) or that models overpredict the magnitude of the effect.

Effects of volcanic emissions (see Chapter 4) on weather have also been suggested. Although spectacular sunsets have been associated with major eruptions (e.g., Krakatoa and Katmai), the effects of scattering of solar radiation from volcanic dust are not clearly understood. Whether a universal effect is created that can result in cooler weather is still at issue. To date, surface temperature and rainfall changes are not conclusively related to volcanic events. The large variability of weather and the self-preserving aspects of the atmospheric system tend to obscure all but the most dramatic changes in climate.

On local scales associated with urban and industrial areas, any significant attenuation of radiation by air pollution can, in addition to other well-recognized factors, result in changes in local weather (e.g., Landsberg, 1970 and 1981). It is possible that local- and regional-scale changes in solar radiation caused by human activity may ultimately influence the heat and water vapor content of the atmosphere on very large scales, but solar radiation and aerosol levels measured at stations remote from pollutant sources have not as yet displayed any trend that can be related to human causes (Fischer, 1967; Ellis and Pueschel, 1971; Hodge et al., 1972).

Unfortunately, there is little agreement about whether the net effect of increased airborne particle concentrations is the warming or cooling of the earth as a whole. Most models can predict either an increase or a decrease in the effective albedo of the earth under cloudless skies, depending on which combination of surface albedo, sun angle, particle size distribution, and particle complex refractive index is assumed. The effects of clouds are very important, and the contributions from infrared radiation must be considered to obtain a complete energy budget (Wesely and Lipschutz, 1976).

9.7 CLOUDINESS AND PRECIPITATION

The global cloud cover plays a vital role in the earth's radiative budget in reflecting energy back to space, in absorbing both solar and longwave (terrestrial) radiation, and in emitting its energy downward and outward into space. Changes in cloud cover, therefore, alter the global heat balance. Cloud- and precipitation-forming processes may be divided into two broad classes: (1) macrophysical processes, which affect the rise and descent of air currents and the amount of water vapor available for condensation; and (2) microphysical processes, which affect the nature of cloud particles formed during condensation. The role of atmospheric aerosols, primarily those that are strongly hygroscopic, is to influence the microphysics of cloud formation.

On a global or even regional scale, the very small amounts of moisture that man adds by land practices or combustion of fossil fuels are negligible in comparison with global evaporation. On a regional scale, only one form of increasing cloudiness suggests itself: the formation of aircraft contrails (Machta and Telegadas, 1974). Aircraft contrail formation results mainly from the injection of water vapor rather than of aerosols.

In urban areas, inadvertent changes of cloudiness as well as the quantity of precipitation have been well established. Such urban impacts also include the frequencies of thunderstorms and hail as well as total amounts of rain. Changnon (1968) has reported a notable increase in days of precipitation, thunderstorms, and hail occurring since 1925 at La Porte, Indiana. Since La Porte is 30 miles east of the Chicago urban-industrial complex, he proposed that the increased precipitation results from inadvertent manmade modifications. Figure 9-39 (Changnon, 1968) shows the 5-year running totals of days with smoke- and haze-restricted visibility in Chicago plotted with 5-year running total precipitation levels at three stations. This measure of atmospheric pollution has a temporal distribution after 1930 rather similar to the La Porte precipitation curve. A noticeable increase in smoke-haze days began

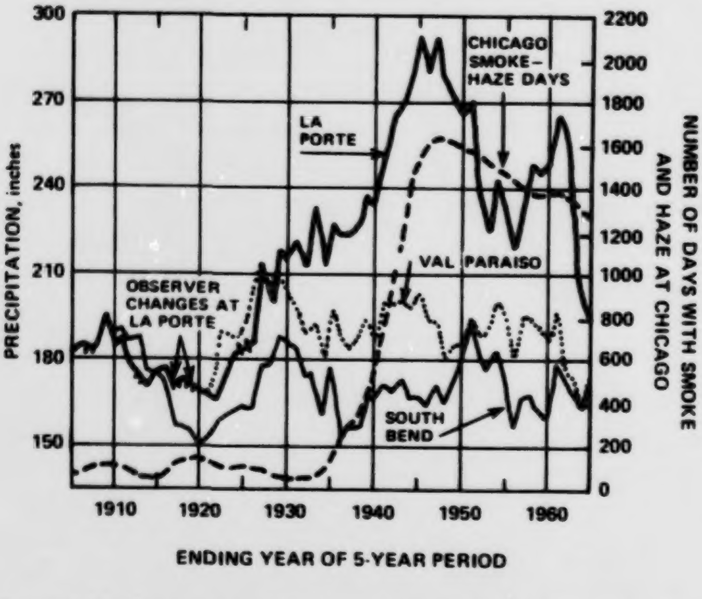


Figure 9-39. Numbers of smoke/haze days are plotted per 5 years at Chicago, with values plotted at end of 5-year period.

Source: Changnon (1968).

in 1935 and became more marked after 1940, when the La Porte precipitation curve began its sharp increase.

A recent report by Clark (1979) critically reexamined this anomaly historically and technically through the use of a budgetary hydrologic model. Clark proposed several explanations of the anomaly. If the anomaly was real, it was, at most, a microscale phenomenom, and its disappearance by the mid-1960s arose from the dissipation of the precipitation-producing mechanism. He noted that although increases in local precipitation have been reported downwind of urban areas, no records exist of any alteration equaling the magnitude of the La Porte precipitation variation relative to surrounding sites. Additionally, the 1929 to 1963 period coincides roughly with the tenure of the sole observer. The accuracy of the gauge was also at issue. Clark concluded that errors by the observer and/or gauge most likely explain the anomaly.

As part of project Metromex, studies by the Illinois State Water Survey suggest increases of about 30 percent in rain and 200 percent in thunderstorms and hail at single gauging stations downwind of St. Louis, with increases of about 10 percent over a two-county area (Changnon et al., 1977; Ackerman et al., 1978). Here again, the physical causes of the maxima are not well understood, but they do appear to be associated with perturbations of the planetary boundary layer and enhanced cloudiness, possibly resulting from the addition of aerosols. It is regrettable that the complex interactions of cloud- and precipitation-forming processes obscure the specific role of manmade aerosols.

The incorporation of particles into rain and fog droplets can change the characteristics of precipitation by changing its chemical composition. The most important impact on precipitation water quality is probably that of "acid rain," discussed in more detail in Chapter 7.

9.8 SUMMARY

Traditionally, visibility has been defined in terms of the distance from an object that is necessary to produce a minimum detectable contrast between that object and its background. Although visibility is often defined by this "visual range," it includes not only being able to see or not see a target, but also seeing targets at shorter distances and appreciating the details of the target, including its colors. Visibility impairment can manifest itself in two ways: (1) as a layer of haze (or a plume), which is visible because it has a visual discontinuity between itself and its background, or (2) as a uniform haze which reduces atmospheric clarity. The type and degree of impairment are determined by the distribution, concentrations, and characteristics of atmospheric particles and gases, which scatter and absorb light traveling through the atmosphere. Scattering and absorption determine light extinction.

On a regional scale, the extinction of light is generally dominated by particle scattering. In urban areas, absorption by particles becomes important and occasionally dominant. Extinction by particles is usually dominated by particles of diameter 0.1 to 2 μm (fine particles). Extinction due to scattering is closely proportional to the fine-particle mass concentration, with extinction/fine mass concentration ratios in the range of about 3 m^2/g for relative humidities below 50-70 percent. For higher humidities, the ratio increases, approximately doubling at 90 percent RH. Extinction/fine mass concentration ratios for absorbing aerosols typically range from 7 to 12 m^2/g.

The currently available visibility monitoring methods measure different aspects of visibility impairment. Generally, contrast-type measurements (such as photography, telephotometry, and human eye observations) relate well to the perception of visual air quality, while extinction or scattering measurements (such as transmissometry and nephelometry) relate to the cause of visibility degradation. Each of the above measurement methods can be used to approximate visual range. No single method is yet widely accepted for measuring light absorption.

Current knowledge indicates that fine particulate matter is composed of varying amounts of sulfate, ammonium, and nitrate ions, elemental carbon, organic carbon compounds, water, and smaller amounts of soil dust, lead compounds, and trace species. Sulfate often dominates the fine mass and light scattering, while elemental C is sometimes the primary visibility-reducing species in urban areas. Significant variations can occur at different times and sites. Our knowledge of the roles of several possibly important species is hindered by the lack of data. The 30-year record of the spatial and temporal trends of coal combustion and visibility suggest that the increasing emissions of SO_x since the 1950's have been associated with similar increases in haziness. Nevertheless, the relationship between SO_2 emissions and resulting sulfate concentrations is not as well known as the relationships between sulfate concentrations and visibility reduction.

Studies performed over the last decade have shown that visibility is a sensitive parameter perceived by the public to indicate polluted air. Loss in the aesthetic value of natural vistas has been ascribed to aerosols of fine PM. A number of studies have been conducted to determine the economic benefit associated with good or improved visibility. The results of these studies, including both contingent and actual market approaches, show that the value people place on visibility is substantial. Although the circumstances of studies differ, the results are broadly consistent, providing evidence that estimation procedures are valid. And, finally, hazards to ground and air transportation have been associated with greatly reduced visibility caused by high concentration of fine PM.

Although the effects on ground transportation from incidents of reduced visibility owing to air pollution are not well documented, they are probably minimal. On the other hand, the effects on aircraft operations are both well documented and significant. Historical records from the National Weather Service indicate that of occasions of visibility of 3 miles (5 km) or less about half occur in the absence of fog, precipitation, or blowing material such as sand or dust. At such low visibility, noninstrument-rated pilots or planes are grounded and commercial air traffic operations at major airports may be significantly reduced.

Pollutants released to the atmosphere alter the environment in ways other than visibility reduction. They may lead to slow and subtle changes in the nature of the atmosphere and, possibly, in climate. For example, a fraction of the solar radiation may be absorbed by aerosols, further reducing the amount of radiation reaching the earth's surface and, at the same time, heating the aerosol layer itself. On a hazy day, the direct solar radiation may be reduced to about one-half of that on a clear day, but most of the energy reappears as diffuse skylight. There is, however, an overall loss of up to about 10 to 20 percent of the radiation reaching the surface.

If there are no clouds between the observer and the sun, the intensity of direct solar radiation for a given solar elevation depends on the variable amount of dust, haze, and water vapor in the atmosphere. The extinction produced by these constituents is called atmospheric turbidity. During hazy episodes, turbidity coefficients as high as 0.6 to 1.0 have been reported, translating into a 75- to 90-percent removal of solar radiation from the direct beam, a 7.5- to 9-percent loss to space, and a 7.5- to 9-percent loss as atmospheric heating. One of the consequences of such a hazy atmosphere is the disappearance of shadow contrast.

The attenuation of solar radiation from scattering and absorption by particles in the atmosphere is probably an important factor in climatic change on all scales. But solar radiation and aerosol levels measured at stations remote from pollutant sources have not, as yet, displayed any trend that can be related to human causes.

Cloud- and precipitation-forming processes may be divided into two broad classes: macrophysical and microphysical processes. Macrophysical processes involve the rise and descent of air masses and the amount of water vapor available for condensation. Atmospheric aerosols, primarily those that are strongly hygroscopic, influence the microphysics of cloud formation. The incorporation of particles into rain and fog droplets can change the characteristics of precipitation by changing its chemical composition. The complex interactions of cloud- and precipitation-forming processes, however, obscure the specific role of manmade aerosols. Accordingly, climatic effects cannot be related quantitatively to pollutant emissions.

9.9 REFERENCES

Ackerman, B., S. A. Changnon, G. Dzurisin, D. F. Eatz, R. C. Grosh, S. P. Hilberg, F. A. Huff, J. W. Mansell, H. G. Ochs, M. E. Peden, P. T. Schickedanz, R. G. Semonin, J. L. Vogel. Summary of Metromex, vol II: Causes of Precipitation Anomalies. Illinois State Water Survey. Bulletin 63, 1978.

Allard, D., and I. Tombach. Evaluation of visibility measurement methods in the eastern United States. Annual Meeting of the Air Pollution Control Association, Montreal, Quebec, June 1980. Paper no. 80-29-3.

Allegrini, I. Optical absorption constant of suspended particulate matter. An air pollution index. Environ. Sci. Tech. 14:1221-1227, 1980.

Allen, J., R. B. Husar, and E. S. Macias. Aerosol size and the shape determination using a laser light scattering spectrometer. In: Aerosol Measurement, a Workshop, University of Florida, Gainesville, Florida, March 24-26, 1976. D. A. Lundgren, M. Lippmann, F. S. Harris, Jr., W. E. Clark, W. H. Marlow, and M. D. Durham, eds., University Presses of Florida, Gainesville, FL, 1979. pp.312-320.

Altshuller, A. P. Atmospheric sulfur dioxide and sulfate, distribution of concentration at urban and nonurban sites in the United States. Environ. Sci. Technol. 7:709-713, 1973.

Angell, J. K., and J. Korshover. Variation in sunshine duration over the contiguous United States between 1950 and 1972. J. Appl. Meteorol. 14: 1174-1181, 1975.

Appel, B. R., E. L. Kothny, E. M. Hoffer, G. M. Hidy, and J. J. Wesolowski. Sulfate and nitrate data from the California Aerosol Characterization Experiment (ACHEX). Environ. Sci. Tech. 12:418-425, 1978.

Appel, B. R., S. M. Wall, Y. Tokiwa, and M. Haik. Interference effects in sampling particulate nitrate in ambient air. Atmos. Environ. 13:319-325, 1979.

Barker, M. L., Planning for environmental indices: observer appraisals of air quality. In: Perceiving Environmental Quality: Research and Applications; Plenum Press, New York, NY, 1976, pp. 175-203.

Barnes, R. A., and D. O. Lee. Visibility in London and the long distance transport of atmospheric sulfur. Atmos. Environ. 12:791-794, 1978.

Bergstrom, R. W. Extinction and absorption coefficients of the atmospheric aerosol as a function of particle size. Beitr. Phys. Atmos. 46:223-224, 1973.

Beuttell, R. G. and A. W. Brewer. Instruments for the measurement of the visual range. J. Sci. Instrum. Phys. Ind. 26:357-359, 1949.

Blackwell, H. R. Contrast thresholds of the human eye. J. Opt. Soc. Am. 36:624-643, 1946.

Bolin, B., and R. J. Charlson. On the role of the tropospheric sulfur cycle in the shortwave radiative climate of the earth. Ambio 5:47-54, 1976.

Booz, Allen and Hamilton, Inc. Study to determine residential soiling costs of particulate air pollution. APTD-0715, U. S. Dept. of Health, Education and Welfare, National Air Pollution Control Administration, Raleigh, NC, October, 1970.

Brookshire, D. S. Issues in valuing visibility: an overview. In: Proceedings of the Workshop in Visibility Values, Fort Collins, CO, January 28-February 1, 1979. D. Fox, R. J. Loomis, and T. C. Green, eds., U.S. Department of Agriculture General Technical Report WO-18, 1979.

Brookshire, D. S., R. C. D'Arge, W. D. Schulze. Methods Development for Assessing Air Pollution Control Benefits, vol. II: Experiments on Valuing Non-Market Goods: A case Study of Alternative Benefit Measures of Air Pollution Control in the South Coast Air Basin of Southern California. EPA-600/5-79-001b, U. S. Environmental Protection Agency, Washington, DC, 1979.

Brookshire, D. S., R. C. D'Arge, W. D. Schulze. Valuing Public Goods: A Comparison of Survey and Hedonic Approaches. Resource and Environmental Economics Laboratory. University of Wyoming, Laramie, WY, 1980.

Brookshire, D. S., B. C. Ives, and W. D. Schulze. The valuation of aesthetic preferences. J. Environ. Manage. 3:325-346, 1976.

Cahill, T. A., B. H. Kusko, L. L. Ashbaugh, J. B. Barone, R. A. Eldred, and E. G. Walther. Regional and local determinations of particulate matter and visibility in the southwestern United States during June and July, 1979. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15:2011-2016, 1981.

Campbell, F. W. and L. Maffei. Contrast and spatial frequency. Sci. Am. 231:106-114, 1974.

Cass, G. R. On the relationship between sulfate air quality and visibility with examples in Los Angeles. Atmos. Environ. 13:1069-1084, 1979.

Changnon, S. A., Jr. The La Porte weather anomaly--fact or fiction? Bull. Am. Meteorol. Soc. 49:4-11, 1968.

Changnon, S. A., F. A. Huff, P. T. Schickedanz, J. L. Vogel. Summary of Metromex, Vol. I: Weather Anomalies and Impacts. Illinois State Water Survey. Bulletin 62, 1977.

Charlson, R. J., H. Horvath, and R. F. Pueschel. The direct measurement of atmospheric light scattering coefficient for studies of visibility and pollution. Atmos. Environ. 1:469-478, 1967.

Charlson, R. J., A. P. Waggoner, J. F. Thielke. Visibility Protection for Class I Areas: The Technical Basis. Council on Environmental Quality, Washington, DC, August 1978a.

Charlson, R. J., D. S. Covert, T. V. Larson, and A. P. Waggoner. Chemical properties of tropospheric sulfur aerosols. Atmos. Environ. 12:39-53, 1978b.

Charlson, R. J., D. S. Covert, Y. Tokiwa, and P. K. Mueller. Multiwavelength nephelometer measurements in Los Angeles, smog aerosol, III. Comparison to light extinction by NO_2. J. Colloid Interface Sci. 39:260-265, 1972.

Charlson, R. J., A. H. Vanderpol, D. S. Covert, A. P. Waggoner, and N. C. Ahlquist. $H_2SO_4/(NH_4)_2SO_4$ background aerosol: optical detection in St. Louis region. Atmos. Environ. 8:1257-1268, 1974.

Clark, R. R. A hydrologic reanalysis of the La Porte anomaly. Bull. Am. Meteorol. Soc. 60: 1979.

Conway, H. M., ed. The Weather Handbook. Conway Research, Inc., Atlanta, GA, 1963.

Cooper, J. A. and J. G. Watson. Portland Aerosol Characterization Study. Presented at the 72d Annual Meeting, Air Pollution Control Association, Cincinnati, OH, June 1979. Paper no. 79-29.4

Cornsweet, T. N. Visual Perception. Academic Press, Inc., New York, NY, 1970.

Countess, R. J., S. H. Cadle, P. J. Groblicki, and G. T. Wolff. Chemical analysis of size-segregated samples of Denver's ambient particulate. GMR-3421, General Motors Research Laboratories, Warren, MI, 1980a.

Countess, R. J., G. T. Wolff, and S. H. Cadle. The Denver winter aerosol: a comprehensive chemical characterization. J. Air Pollut. Control Assoc. 30:1194-1200, 1980b.

Covert, D. S., R. J. Charlson, and N. C. Ahlquist. A study of the relationship of chemical composition and humidity to light scattering by aerosols. J. Appl. Meteorol. 11:968-976, 1972.

Covert, D. S., A. P. Waggoner, R. E. Weiss, N. C. Ahlquist, and R. J. Charlson. Atmospheric aerosols, humidity, and visibility. Adv. Environ. Sci. Technol. 9:559-581, 1980.

Crosby, P., and B. W. Koerber. Scattering of light in the lower atmosphere. J. Opt. Soc. Am. 53:358-361, 1963.

Cwalinski, R., J. M. Lansinger, and W. G. Tank. Field testing and evaluation of methods for measuring visibility. EPA-650/2-75-039, U.S. Environmental Protection Agency, Research Triangle Park, N.C., 1975.

Draftz, R. G. Identification and sources of Denver aerosol. Presented at the 67th Annual Meeting, Air Pollution Control Association, 1974. Paper no. 74-263.

Duntley, S. Q. Visibility, II. Summary. Appl. Opt. 3:551-556, 1964.

Edwards, J. D. A comparison of the British smoke shade method and the integrating plate method. Master's Thesis, University of Washington, Seattle, WA, 1980.

Eiden, R. Determination of the complex index of refraction of spherical aerosol particles. Appl. Opt. 10:749-754, 1971.

Eldred, R. A., L. L. Ashbaugh, T. A. Cahill, R. G. Flocchini, and M. L. Pitchford. The effect of the 1980 smelter strike on air air quality in the southwest. Report #AQG 81-025, Air Quality Group, Crocker Nuclear Laboratory, Univ. of California at Davis, Davis, CA, 1981.

Ellenson, W. D., R. C. Nininger, E. T. Myers, J. H. Overton, Jr., E. W. Corse, J. C. King, D. B. Kirkton, and P. L. Smith. Data report - visibility study in the Great Smoky Mountains region - September 1980. U.S. Environmental Protection Agency Contract 68-02-2566, Northrop Services, Inc., Research Triangle Park, NC, 1981.

Ellestad, T. G. Aerosol composition of urban plumes passing over a rural monitoring site. Ann. N. Y. Acad. Sci. 338:202-218, 1980.

Ellestad, T. G., and R. E. Speer. Application of a telephotometer to visibility measurements in the eastern United States. Atmos. Environ 15:2443-2449, 1981.

Ellis, A. T., and R. F. Pueschel. Absence of air pollution trends at Mauna Loa. Science 172:845-846, 1971.

Elterman, P. Integrating cavity spectroscopy. Appl. Opt. 9:2140-2142, 1970.

Ensor, D., and A. P. Waggoner. Angular truncation error in the integrating nephelometer. Atmos. Environ. 4:481-487, 1970.

Ensor, D. S., R. J. Charlson, N. C. Ahlquist, K. T. Whitby, R. B. Husar, and B. Y. H. Liu. Multiwavelength nephelometer measurements in Los Angeles smog aerosol. I. Comparison of calculated and measured light scattering. J. Colloid Interface Sci. 39:242-251, 1972.

Faxvog, F. R. Optical scattering per unit mass of single particles. Appl. Opt. 14:269-270, 1975.

Faxvog, F. R., and D. M. Roessler. Carbon aerosol visibility vs. particle size distribution. Appl. Opt. 17:2612-2616, 1978.

Federal Aviation Administration. Airmen's information manual. Basic Flight Information - ATC Procedures, January, 1978a.

Federal Aviation Administration. Air Traffic Service Performance measurement system for major airports. Nov. 1975-1980. U.S. Dept. of Transportation, Washington, DC, 1980a.

Federal Aviation Administration. Census of U.S. Civil Aircraft. U.S. Dept. of Transportation, Washington, DC, 1980b.

Federal Aviation Administration. General Aviation Avionics Statistics. U.S. Dept. of Transportation, Washington, DC, April 1981a. p. 19.

Federal Aviation Administration. U.S. Civil Airmen Statistics. U.S. Dept. of Transportation, Washington, DC, 1981b.

Ferman, M. A., G. T. Wolff, and N. A. Kelly. The nature and sources of haze in the Shenandoah Valley/Blue Ridge Mountains area. J. Air Pollut. Control Assoc. 31:1074-82, 1981.

Fischer, W. H. Some atmospheric turbidity measurements in Antarctica. J. Appl. Meteorol. 6:958-959, 1967.

Flachsbart, P. G., and S. Phillips. An index and model of human response to air quality. J. Air Pollut. Control Assoc. 30:759-768, 1980.

Flowers, E. C., R. A. McCormick, and K. R. Kurfis. Atmospheric turbidity over the United States, 1961-1966. J. Appl. Meteorol. 8:955-962, 1969.

Fowler, B. W., and C. C. Sung. Scattering of an electromagnetic wave from dielectric bodies of irregular shape. J. Opt. Soc. Am. 69:756-761, 1979.

Fox, D., R. J. Loomis, and T. C. Green, eds. Proceedings of the workshop in Visibility Values, Fort Collins, CO, January 28-February 1, 1979. U.S. Dept. of Agriculture General Technical Report WO-18.

Gates, D. M. Spectral distribution of solar radiation at the earth's surface. Science 151:523-529, 1966.

Glass, N. R. (ed.) Environmental Effects of Increased Coal Utilization: Ecological Effects of Gaseous Emissions from Coal Combustion. EPA-600/7-78-108, U.S. Environmental Protection Agency, Research Triangle Park, NC, 1978.

Grams, G. W., I. H. Blifford, Jr., D. A. Gillette, and P. B. Russell. Complex index of refraction of airborne soil particles. J. Appl. Meteorol. 13:459-471, 1974.

Green, D. M., and F. W. Campbell. Optical and retinal factors affecting visual resolution. J. Physiol. 181:576-593, 1965.

Griffing, G. W. Relations between the prevailing visibility, nephelometer scattering coefficient, and sunphotometer turbidity coefficient. Atmos. Environ. 14:577-584, 1980.

Groblicki, P. J., G. T. Wolff, and R. J. Countess. Visibility reducing species in the Denver brown cloud. Part 1: Relationships between extinction and chemical composition. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15: 2473-2484, 1981.

Grosjean, D., G. J. Doyle, T. M. Mischke, M. P. Poe, D. R. Fitz, J. P. Smith, and J. N. Pitts, Jr. Concentration, size distribution, and modes of formulation of particulate nitrate, sulfate and ammonium compounds in the eastern part of the Los Angeles air basin. Presented at the 69th Annual Meeting, Air Pollution Control Association, 1976. Paper No. 76-20.3.

Hall, F. F., Jr. Visibility reductions from soil dust in the western U.S. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15:1929-1933, 1981.

Hall, J. S., and L. A. Riley. Basic spectrophotometric measures of air quality over long paths. In: Radiative Transfer and Thermal Control, vol. 49: Progress in Astronautics and Aeronautics. A. M. Smith, ed., American Institute of Aeronautics and Astronautics, New York, NY, 1976. pp. 205-212.

Hänel, G. New results concerning the dependence of visibility on relative humidity and their significance in a model for visibility forecast. Beitr. Phys. Atmos. 44:137-167, 1971.

Hänel, G. The properties of atmospheric aerosol particles as functions of the relative humidity at thermodynamic equlibrium with the surrounding moist air. In: Advances in Geophysics. Landsberg, H. E. and J. van Mieghen, eds. Academic Press, New York, NY, 1976.

Harrison, A. W. Nephelometer estimates of visual range. Atmos. Environ. 13:645-652, 1979.

Heintzenberg, J., and Quenzel, H. On the effect of the loss of large particles on the determination of scattering coefficients with integrating nephelometers. Atmos. Environ. 7: 503-507, 1973.

Heisler, S. L., R. C. Henry, and J. G. Watson. The sources of the Denver Haze in November and December of 1978. Presented at the 73d Annual Meeting, Air Pollution Control Association, 1980. Paper no. 80-58.6.

Henry, R. C. Psychophysics and visibility values. In: Proceedings of the Workshop in Visibility Values, Fort Collings, CO, January 28-February 1, 1979. D. Fox, R. J. Loomis, and T. C. Greene, U.S. Dept. of Agriculture, General Technical Report WO-18.

Ho, W., Hidy, G. M., and Govan, R. M. Microwave measurements of the liquid water content of atmospheric aerosols. J. Appl. Meteorol. 13:871-879, 1974.

Hobbs, P. V., H. Harrison, and E. Robinson. Atmospheric effects of pollutants. Science 183:909-915, 1974.

Hodge, P. W., N. Laulainen, and R. J. Charlson. Astronomy and air pollution. Science 178:1123-1124, 1972.

Hodkinson, J. R. Calculations of colour and visibility in urban atmospheres polluted by gaseous NO_2. Air Water Pollut. 10:137-144, 1966.

Hoffmann, H. E., and Kuehnemann, W. Comparison of the results of two measuring methods determining the horizontal standard visibility with the visual visibility range. Atmos. Environ. 13:1629-1634, 1979.

Horvath, H., and K. E. Noll. The relationship between atmospheric light scattering coefficient and visibility. Atmos. Environ. 3:543-550, 1969.

Husar, R. B., and D. E. Patterson. Regional scale air pollution: sources and effects. Ann. N.Y. Acad. Sci. 338:399-417, 1980.

Husar, R. B., and J. M. Holloway. Visibility trend at Blue Hill, ME since 1889. Bull. of Am. Meteorol. Soc. in press, 1981.

Husar, R. B., D. E. Patterson, J. M. Holloway, W. E. Wilson, and T. G. Ellestad. Trends of eastern U.S. haziness since 1948. In: Proceedings of the Fourth Symposium on Atmospheric Turbulence, Diffusion, and Air Pollution, American Meteorological Society, Reno, NE, 1979. pp. 249-256.

Janzen, J. Extinction of light by highly non-sperical strongly absorbing colloidal particles: spectrophotometric determination of volume distributions for carbon blacks. Appl. Opt. 19:2977-2985, 1980.

Jennings, S. G., and Pinnick, R. G. Relationships between visible extinction, absorption and mass concentration of carbonaceous smokes. Atmos. Environ. 14:1123-1129, 1980.

Kerker, M. The Scattering of Light and Other Electromagnetic Radiation. Academic Press, Inc., New York, NY, 1969.

Koschmieder, H. Theorie der horizontalen Sichtweite. [theory of the horizontal visual range.] Beitr. Phys. Freien Atmos. 12:33-53, 171-181, 1924.

Landsberg, H. E. Man-made climatic changes. Science 170:1265-1274, 1970.

Landsberg, H. E. The Urban Climate, International Geophysical Series, Vol. 28, Academic Press, New York, 1981.

Latimer, D. A., R. W. Bergstrom, S. R. Hayes, M. K. Liu, J. H. Seinfeld, G. Z. Whitten, M. A. Wojcik, and M. J. Hillyer. The Development of Mathematical Models for the Prediction of Anthropogenic Visibility Impairment. EPA-450/3/78-110a, U.S. Environmental Protection Agency, Research Triangle Park, NC, September 1978.

Latimer, P. Predicted scattering by spheroids: comparison of approximate and exact methods. Appl. Opt. 19:3039-3041, 1980.

Lawson, D. R., and J. W. Winchester. Atmospheric sulfur aerosol concentrations and characteristics from the South American Continent. Science 205:1267-1269, 1979.

Leaderer, B. P., D. M. Bernstein, J. M. Daisey, M. T. Kleinman, T. J. Kneip, E. O. Knutson, M. Lippmann, P. J. Lioy, K. A. Rahney, D. Sinclair, R. L. Tanner, G. T. Wolff. Summary of the New York summer aerosol study (NYSAS). J. Air Pollut. Control Assoc. 28:321-327, 1978.

Leonard, E. M., M. D. Williams, and J. P. Mutschlecner. The visibility issue in the Rocky Mountain West. Prepared by Los Alamos Scientific Laboratory for the Dept. of Energy, preliminary draft report, Sept. 30, 1977.

Lewis, C. W., and E. S. Macias. Composition of size-fractioned aerosol in Charleston, West Virginia. Atmos. Environ. 14:185-194, 1980.

Lin, C. I., M. Baker, and R. J. Charlson. Absorption coefficient of atmospheric aerosol: a method for measurement. Appl. Opt. 12:1356-1363, 1973.

Lindberg, J. D., and L. S. Laude. Measurement of the absorption coefficient of atmospheric dust. Appl. Opt. 13:1923-1927, 1974.

Machta, L., and K. Telegadas. Inadvertent large-scale weather modification. In: Weather and Climate Modification. W. N. Hess, ed., John Wiley & Sons, New York, NY, 1974. pp. 687-725.

Macias, E. S., and R. B. Husar. A review of atmospheric particulate mass measurement via the beta attenuation technique. In: Fine Particles, Aerosol Generation Measurement, Sampling, and Analysis. B. Y. H. Liu, ed., Academic Press, New York, NY, 1976.

Macias, E. S., D. L. Blumenthal, J. A. Anderson, and B. K. Cantrell. Size and composition of visibility-reducing aerosols in southwestern plumes. Ann. N.Y. Acad. Sci. 338:233-257, 1980.

Macias, E. S., J. O. Zwicker, J. R. Ouimette, S. V. Hering, S. K. Friedlander, T. A. Cahill, G. A. Kuhlmey, and L. W. Richards. Regional haze case studies in the southwestern U.S.: I. Aerosol chemical composition. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15:1971-1986, 1981a.

Macias, E. S., J. O. Zwicker, and W. H. White. Regional haze case studies in the southwestern U.S. - II. Source contributions. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15:1987-9997, 1981b.

Malm, W. Visibility: A physical perspective. In: Proceedings of the Workshop in Visibility Values. Fort Collins, Colorado, January 28-February 1, 1979. D. Fox, R. J. Loomis, and T. C. Greene, eds. U.S. Dept. of Agriculture General Technical Report WO-18, 1979a. pp. 56-68.

Malm, W. C. Considerations in the measurement of visibility. J. Air Pollut. Control Assoc. 29:1042-1052, 1979b.

Malm, W. C. K. K. Kelly, and J. V. Molenar. Human perception of visual air quality (Uniform Haze). In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15:1875-1890, 1981a.

Malm, W. C., E. G. Walther, K. E. O'Dell and M. Kleine. Visibility in the Southwestern United States from summer 1978 to spring 1979. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15:2031-2042, 1981b.

Malm, W., A. Pitchford, R. Tree, E. Walther, M. Pearson, and S. Archer. The visual air quality predicted by conventional and scanning teleradiometers and an integrating nephelometer. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos Environ. 15:2547-2554, 1981c.

Manabe, F., and R. Wetherald. The effects of doubling the CO_2 concentration on the climate of a general circulation model. J. Atmos. Sci. 32:3-15, 1975.

Marians, M., and J. Trijonis. Empirical Studies of the Relationship Between Emissions and Visibility in the Southwest. Prepared under grant 802015 by Technology Service Corp. U.S. Environmental Protection Agency, Cincinnati, OH, 1979.

McCree, K. J., and M. E. Keener. Effect of atmospheric turbidity on the photosynthetic rates of leaves. Agric. Meteorol. 13:349-357, 1974.

Middleton, W. E. K. The colors of distant objects. J. Opt. Soc. Am. 40:373-376, 1950.

Middleton, W. E. K. Vision Through the Atmosphere. University of Toronto Press, Toronto, Canada, 1952.

Mie, G. Optics of cloudy media, especially colloidal metal sols. Ann. Phys. 25:377-445, 1908.

Mugnai, A., and W. J. Wiscombe. Scattering of radiation by moderately nonspherical particles. J. Atmos. Sci. 37:1291-1307, 1980.

National Center for Air Pollution Control. New York - New Jersey Air Pollution Abatement Activity. Particulate Matter. Phase II. Pre-Conference Investigations. U.S. Department of Health, Education, and Welfare, Public Health Service, Cincinnati, OH, December, 1967. p.105.

National Park Service. My Eyes Need a Good Stretching; Seven Authorities Speak Out on Visibility -- Clean Air and Unique Natural Areas. U.S. Department of the Interior, Washington, DC, 1978.

National Transportation Safety Board. Annual review of aircraft accident data. NTSB-ARC-80-1 Washington, D. C. 1978a.

National Transportation Safety Board. Briefs of Fatal Accidents Involving Weather as a Cause/Factor, U.S. General Aviation, 1978. NTSB-AMM-80-5, National Transportation Safety Board, Washington, DD, 1978b.

National Weather Service, Federal Meteorological Handbook No. 1, Surface Observations. National Oceanographic and Atmospheric Administration, Washington, DC, 1979.

National Weather Service. Historical data base of visibility measurements for 147 U. S. sites 1951-1980. National Oceanographic and Atmospheric Administration, Washington, DC, 1981.

Nininger, R. C., J. H. Overton, Jr., M. J. Kaiserman, V. P. Aneja, E. W. Corse, D. B. Kirkton, and E. T. Myers. Data report: visibility study in the Ohio River basin - February 1980. U.S. EPA Contract 68-02-2566. Northrop Services, Inc. Research Triangle Park, NC, 1981a.

Nininger, R. C., W. D. Ellenson, J. H. Overton, Jr., R. A. Carlson, W. S. Crews, J. W. Faircloth, J. C. King, E. T. Myers, P. L. Smith, M. J. Turner, and M. L. Wheeler. Data report: visibility study in the Ohio River basin - July 1980. U.S. EPA Contract 68-02-2566. Northrop Services, Inc. Research Triangle Park, NC, 1981b.

Orel, A. E., and J. H. Seinfeld. Nitrate formation in atmospheric aerosols. Environ. Sci. Tech. 11:1000-1007, 1977.

Orgill, M. M., and G. A. Sehmel. Frequency and diurnal variation of dust storms in the contiguous United States. Atmos. Environ. 10:813-825, 1976.

Orr, C., F. K. Hurd, and W. J. Corbett. Aerosol size and relative humidity. J. Colloid Interface Sci. 13:472-482, 1958.

Ouimette, J. R. Aerosol Chemical Species Contributions to the Extinction Coefficient. Ph.D. Thesis, California Institute of Technology, Pasadena, CA, 1980.

Pashel, G. E., and D. R. Egner. A comparison of ambient suspended particulate matter concentrations as measured by the British smoke sampler and the high volume sampler at 16 sites in the United States. Atmos. Environ. 15:919-927, 1981.

Patel, A. S. Spatial resolution by the human visual system. The effect of mean retinal illuminance. J. Opt. Soc. Am. 56:689-694, 1966.

Patterson, E. M. Atmospheric extinction between 0.55 mm and 10.6 mm due to soil-derived aerosols. Appl. Opt. 16:2414-2418, 1977.

Patterson, E. M., and D. A. Gillette. Measurements of visibility vs. mass-concentration for airborne soil particles. Atmos. Environ. 11:193-196, 1977.

Patterson, R. K., and J. Wagman. Mass and composition of an urban aerosol as a function of particle size for several visibility levels. J. Aerosol Sci. 8:269-279, 1977.

Penndorf, R. Tables of the refractive index for standard air and the Rayleigh scattering coefficient for the spectral region between 0.2 and 20.0 m and their application to atmospheric optics. J. Opt. Soc. Am. 47:176-182, 1957.

Peterson, J. T., and E. C. Flowers. Interactions between air pollution and solar radiation. Sol. Energy 19:23-32, 1977.

Pierson, W. R., and P. B. Russell. Aerosol carbon in the Denver area in November 1973. Atmos. Environ. 13:1623-1628, 1979.

Pierson, W. R., W. W. Brachaczek, T. J. Truex, J. W. Butler, and T. J. Korniski. Ambient sulfate measurements on Alleghany Mountains and the question of atmospheric sulfate in the northeastern United States. Ann. N.Y. Acad. Sci. 338:145-173, 1980.

Pinnick, R. G., D. E. Carroll, and D. J. Hofmann. Polarized light scattered from monodisperse randomly oriented nonspherical aerosol particles: measurements. Appl. Opt. 15:384-393, 1976.

Porch, W. M., R. J. Charlson, and L. F. Radke. Atmospheric aerosol: does a background level exist? Science 170:315-317, 1970.

Presle, G. and R. Pirich. The relation between changes of the extinction coefficient and relative humidity. J. Aerosol Sci. 11:255-257, 1980.

Pueschel, R. F. ahd P. A. Allee. Visibility, climatology, and air quality in east-central Utah. Presented at the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980.

Pueschel, R. F., and D. L. Wollman. On the nature of atmospheric background aerosol. Presented at the 14th Conference on Agricultural and Forest Meteorology, Minneapolis, MN, April 1978.

Rabinoff, R., and B. Herman. Effect of aerosol size distribution on the accuracy of the integrating nephelometer. J. Appl. Meteorol. 12:184-186, 1973.

Randall, A., B. Ives, C. Eastman. Bidding games for valuation of aesthethic environmental improvements. J. Environ. Econ. Manage. 1:132-149, 1974.

Robinson, G. D. Absorption of solar radiation by atmospheric aerosol, as revealed by measurements at the ground. Arch. Meteorol. Geophys. Bioclimatol. Ser. B 12:19, 1962.

Roessler, D. M., and F. R. Faxvog. Optical properties of agglomerated acetylene smoke particles at 0.5145 μm and 10.6 μm wavelengths. J. Opt. Soc. Am. 70:230-235, 1980.

Rosen, H., A. D. A. Hansen, R. L. Dod, and T. Novakov. Soot in urban atmospheres: determination by an optical absorption technique. Science 208:741-744, 1980.

Rosen, H., A. D. A., Hansen, L., Grundel, and T. Novakov. Identification of the optically absorbing component in urban aerosols. Appl. Opt. 17:3859-3861, 1978.

Rowe, R. D., and L. G. Chestnut. Visibility Benefits Assessment Guidebook. EPA-450/5-81-001, U.S. Environmental Protection Agency, Research Triangle Park, NC, August 1981.

Rowe, R. D., R. C. d'Arge, and D. S. Brookshire. An experiment on the economic value of visibility. J. Environ. Econ. Manage. 7:1-19, 1980a.

Rowe, R. D., R. C. d'Arge, and D. S. Brookshire. Environmental Preference and Effluent Charges. In: Progress in Resource Management and Environmental Planning. Timothy O'Riordin and R. Kerry Turner, eds. London, England. John Wiley & Sons, Ltd. 1980b. (Series in Environmental Economics)

Ruby, M. G., and A. P. Waggoner. Intercomparison of integrating nephelometer measurements. Environ. Sci. Tech. 15:109-113, 1981.

Ruppersberg, G. H. Registrierung der sichtweite mit dem streulichtschreiber. [Reading of the visual range with a light scattering recorder.] Beitr. Phys. Atm. 37:252-263, 1964.

Sadler, M., R. J. Charlson, H. Rosen, and T. Novakov. An intercomparison of the integrating plate and the laser transmission methods for determination of aerosol absorption coefficients. Atmos. Environ. 15:1265-1268, 1981.

Samuels, H. J., S. A. Twiss, E. W. Wong, H. Wong-Woo, P. K. Mueller, J. A. Maga, and G. S. Taylor. Visibility, Light Scattering and Mass Concentration of Particulate Matter. Report of the California Tri-City Aerosol Sampling Project. State of California Air Resources Board, Sacramento, CA, July 1973.

Schober, H. A. W., and R. Hilz. Contrast sensitivity of the human eye for square-wave grating. J. Opt. Soc. Am. 55:1086-1091, 1965.

Schusky, J. Public Awareness and Concern with Air Pollution in the St. Louis Metropolitan Area, J. Air Pollut. Control Assoc., 16:72-76, 1966.

Schuerman, D. W., ed. Light Scattering by Irregularly Shaped Particles. Plenum, New York, NY, 1980.

Shah, J. J., T. L. Johnson, E. K. Heyerdahl, and J. J. Huntzicker. Carbonaceous aerosol at urban and rural sites in the United States. Presented at the 75th annual meeting of the Air Pollution Control Association, New Orleans, June 1982. Paper 82-65.5.

Sloane, C. S. Visibility trends II. Mideastern United States 1948-1978. Report no. 10, General Motors Research Laboratories, Warren, MI, 1980.

Snedecor, G. W., and W. G. Cochran. Statistical methods. Iowa State University Press, Ames, IO, 1980.

Spicer, C. W., and P. M. Schumacher. Interferences in sampling atmospheric particulate nitrate. Atmos. Environ. 11:873-876, 1977.

Spicer, C. W., and P. M. Schumacher. Particulate nitrate: laboratory and field studies of major sampling interferences. Atmos. Environ. 13:543-552, 1979.

Steffens, C. Measurement of visibility by photographic photometry. Ind. Eng. Chem. 41:2396-2399, 1949.

Stevens, R. K., T. G. Dzubay, G. Russworm, and D. Rickel. Sampling and analysis of atmospheric sulfates and related species. Atmos. Environ. 12:55-68, 1978.

Stevens, R. K., T. G. Dzubay, R. W. Shaw, W. A. McClenny, C. W. Lewis, and W. E. Wilson. Characterization of aerosol in the Great Smoky Mountains. Environ. Sci. Tech. 14:1491-1498, 1980.

Stuiver, M. Atmospheric carbon dioxide and carbon reservoir changes. Science 199:253-258, 1980.

Stukel, J. J., and B. R. Keenan. Ohio River Basin Energy Study Phase I: Interim Findings. EPA-600/7-77-120. U.S. Environmental Protection Agency, Research Triangle Park, NC, 1977.

Tang, I. N. Deliquescence properties and particle size change of hygroscopic aerosols. In: Generation of Aerosols and Facilities for Exposure Experiments. K. Willeke, ed., Ann Arbor Science, Ann Arbor, MI, 1980.

Tang, I. N., H. R. Munkelwitz, and J. G. Davis. Aerosol growth studies, IV. Phase transformation of mixed salt aerosols in a moist atmosphere. J. Aerosol Sci. 9:505-511, 1978.

Tanner, R. L., R. Garber, W. Marlow, B. P. Leaderer,and M. A. Leyko. Chemical composition of sulfate as a function of particle size in New York summer aerosol. Ann. N. Y. Acad. Sci. 322:99-113, 1979.

Taylor, J. H. Use of visual performance data in visibility prediction. Appl. Opt. 3:562-569, 1964.

Trijonis, J., and R. Shapland. Existing Visibility Levels in the U.S.: Isopleth Maps of Visibility in Suburban/Nonurban Areas During 1974-1976. EPA-45C/5-79-010, U.S. Environmental Protection Agency, Research Triangle Park, NC, 1979.

Trijonis, J., and K. Yuan. Visibility in the Southwest: An Exploration of the Historical Data Base. EPA-600/3/78/039, U.S. Environmental Protection Agency, Research Triangle Park, NC, April 1978a.

Trijonis, J., and K. Yuan. Visibility in the Northeast: Long Term Visibility Trends and Visibility/Pollutant Relationships. EPA-600/3-78-075, U.S. Environmental Protection Agency, Research Triangle Park, NC, August 1978b.

Truex, T. J., and J. E. Anderson. Mass monitoring of carbonaceous aerosols with a spectrophone. Atmos. Environ. 13:507-509, 1979.

Twomey, S. Pollution and the planetary albedo. Atmos. Environ. 8:1251, 1974.

U.S. Bureau of Mines. Minerals Yearbook, Annual Publication, 1933-74. U.S. Department of the Interior, Washington, D.C.

U.S. Congress. Senate. Committee on Public Works. A Study of Pollution - Air. U.S. Government Printing Office, Washington, D.C. September, 1963. p. 21.

U.S. Department of Commerce. Climatic Atlas of the United States. U.S. Dept. of Commerce, Washington, D.C., 1968.

U.S. Department of Energy. Carbon Dioxide Effects, Research, and Assessment Program. A Comprehensive Plan for Carbon Dioxide Effects, Research, and Assessment, Part I: The Global Carbon Cycle and Climatic Effects of Increasing Carbon Dioxide. DOE/EV-0094, U.S. Department of Energy, Washington, DC, 1980.

U.S. Department of Energy. Office of Carbon Dioxide Effects, Research, and Assessment. A Comprehensive Plan for Carbon Dioxide Effects, Research, and Assessment. U.S. Department of Energy, Washington, DC, 1978.

U.S. Dept. of Health, Education and Welfare. Public Awareness and Concern with Air Pollution in the St. Louis Metropolitan Areas. U.S. Dept. of Health, Education and Welfare, Div. of Air Pollution, Washington, DC, 1965.

U.S. Environmental Protection Agency. EPA Report to Congress: Protecting Visibility. EPA-450/ 5-79-008, U. S. Environmental Protection Agency, Research Triangle Park, NC, 1979.

Van Nes, F. L., and M. A. Bouman. Spatial modulation transfer in the human eye. J. Opt. Soc. Am. 57:401-406, 1967.

Veress, S. A. Extinction coefficient. Photo. Eng. 40:183-191, 1972.

Waggoner, A. P., M. B. Baker, and R. J. Charlson. Optical absorption by atmospheric aerosols. Appl. Opt. 12:896, 1973.

Waggoner, A. P., and R. J. Charlson. Measurement of aerosol optical parameters. In: Fine Particles: Aerosol Generation Measurement, Sampling, and Analysis. B. Y. H. Liu, ed., Academic Press, Inc., New York, NY, 1976.

Waggoner, A. P., and R. E. Weiss. Comparisons of fine particle mass concentration and light scattering extinction in ambient aerosol. Atmos. Environ. 14:623-626, 1980.

Waggoner, A. P., R. E. Weiss, and T. V. Larson. In situ, rapid response measurement of $H_2SO_4/(NH_4)_2SO_4$ aerosols in urban Houston: a comparison with rural Virginia. Submitted to Atmospheric Environment, August 1982.

Waggoner, A. P., R. E. Weiss, N. C. Ahlquist, D. S. Covert, S. Will, and R. J. Charlson. Optical characteristics of atmospheric aerosols. Atmos. Environ. 15:1891-1909, 1981.

Wall, G. Public response to air pollution in South Yorkshire, England. Environ. Behav. 5:219-248, 1973.

Weil, H., and C. M. Chu. Scattering and absorption by thin flat aerosols. Appl. Opt. 19:2066-2071, 1980.

Weiss, R. E. The Optical Absorption Properties of Suspended Particles in the Lower Troposphere at visible wavelengths. Ph.D. Thesis, University of Washington, 1980.

Weiss, R. E., and A. P. Waggoner. The importance of aerosol absorption and graphitic carbon in visibility and atmospheric optics. Presented at the 74th Annual Meeting, Air Pollution Control Association, Philadelphia, June 1981. Paper 81-54-6.

Weiss, R. E., T. V. Larson, and A. P. Waggoner. In situ, rapid response measurement of $H_2SO_4/(NH_4)_2$ SO_4 aerosols in rural Virginia. Env. Sci. Tech. 16:525-532, 1982.

Weiss, R. E., A. P. Waggoner, R. J. Charlson, and N. C. Ahlquist. Sulfate aerosol: its geographical extent in the midwestern and southern United States. Science 195:979-981, 1977.

Weiss, R. E., A. P. Waggoner, R. J. Charlson, D. L. Thorsell, J. S. Hall, and L. A. Riley. Studies of the optical, physical, and chemical properties of light-absorbing aerosols. In: Proceedings of the Conference on Carbonaceous Particles in the Atmosphere, March 20-22, 1978. LBL-9037, T. Novakov, Ed., Laurence Berkeley Laboratory, University of California, 1979. p. 257-262.

Wesely, M. L., and R. C. Lipschutz. An experimental study of the effects of aerosols on diffuse and direct solar radiation received during the summer near Chicago. Atmos. Environ. 10:981-987, 1976.

Whitby, K. T., and G. M. Sverdrup. California aerosols: their physical and chemical characteristics. Adv. Environ. Sci. Tech. 9:477-517, 1980.

White, W. H., and D. E. Patterson. Nitrogen dioxide, particles, and the color of smoke plumes. In: Proceedings of the Symposium on Plumes and Visibility: Measurements and Model Components, Grand Canyon National Park, AZ, November 10-14, 1980. Atmos. Environ. 15:2097-2104, 1981.

White, W. H., and P. T. Roberts. On the nature and origins of visibility reducing aerosols in the Los Angeles Air Basin. Atmos. Environ. 11:803-812, 1977.

Williams, J. D., and F. L. Bunyard. Interstate Air Pollution Study, Phase II Project report, Vol. VII - Opinion Surveys and Air Quality Statistical Relationships. U.S. Dept. of Health, Education and Welfare, Div. of Air Pollution, Cincinnati, OH, 1966.

WMO/EPA/NOAA/UNEP. Global Monitoring of the Environment for Selected Atmospheric Constituents 1972-1975. Environmental Data Service National Climatic Center, Asheville, NC, 1974-1977.

Wolff, G. T., M. A. Ferman, N. A. Kelly, D. P. Stroup, and M. S. Ruthkosky. The relationships between the chemical composition of fine particles and visibility in the Detroit metropolitan area. GMR-4072, ENV # 128, General Motors Research Laboratories, Warren, MI, 1982.

Wolff, G. T., N. A. Kelly, M. A. Ferman. On the sources of summertime haze in the eastern United States. Science 211:703-705, 1981.

Wolff, G. T., P. J. Groblicki, S. H. Cadle, and R. J. Countess. Particulate carbon at various locations in the United States. Presented at the General Motors Symposium on Particulate Carbon, Warren, MI, October 1980.

10. EFFECTS ON MATERIALS

10.1 INTRODUCTION

Beginning with associative 17th century observations and continuing through modern analytical investigation, scientists have gathered evidence that air pollutants damage man-made and natural materials. Pollutant-related damage may lead to measures such as increased maintenance, use of protective coatings, replacement of materials, or the control of emissions. The decision to pursue one course rather than another is guided, ultimately, by weighing the damage against the costs of prevention. Neither damage nor prevention cost is necessarily measured monetarily, especially for works of art or items of historical significance. For better or worse, however, society tends to assign a monetary value to value gained or lost. This chapter presents and evaluates studies useful for estimating the dollar cost of material damage caused by sulfur oxides and particulate matter.

Figure 10-1 depicts the relation between pollutant emissions and economic damage. As shown, one may (1) proceed from ambient pollutant levels to economic damage estimates directly or (2) estimate damage based on physical damage functions. The latter route, called the damage function approach, has been the preferred method, although more recent studies have employed the first route. The estimation of willingness to pay is common to both choices.

Economic damage (benefit) that results from increased (decreased) pollutant concentrations can be estimated by willingness-to-pay approaches. All willingness-to-pay approaches try to estimate the aggregate monetary values that all affected individuals assign to the effects of a change in pollutant concentration. These approaches can be divided into three classes: damage function approaches, nonmarket approaches, and indirect market approaches. The first step of the damage function approach uses the relationship of pollutant exposure to physical damage. The second step links the physical damage to a dollar estimate of willingness to pay. Most economic damage estimates using this approach have not considered substitution possibilities for producers or consumers; however, proper consideration of these factors can yield good estimates of willingness to pay (via the damage function approach). Nonmarket approaches generally use surveys which attempt to ascertain the monetary values assigned to the effects. Indirect market approaches use information about the demand for marketed goods to estimate the willingness to pay for nonmarketed environmental attributes that are closely related to the marketed good (e.g., property value studies that estimate the willingness to pay for a change in the level of pollutant concentration through analyses of the changes in price of residential property) (Freeman, 1979a). Each of these three approaches has a different data requirement.

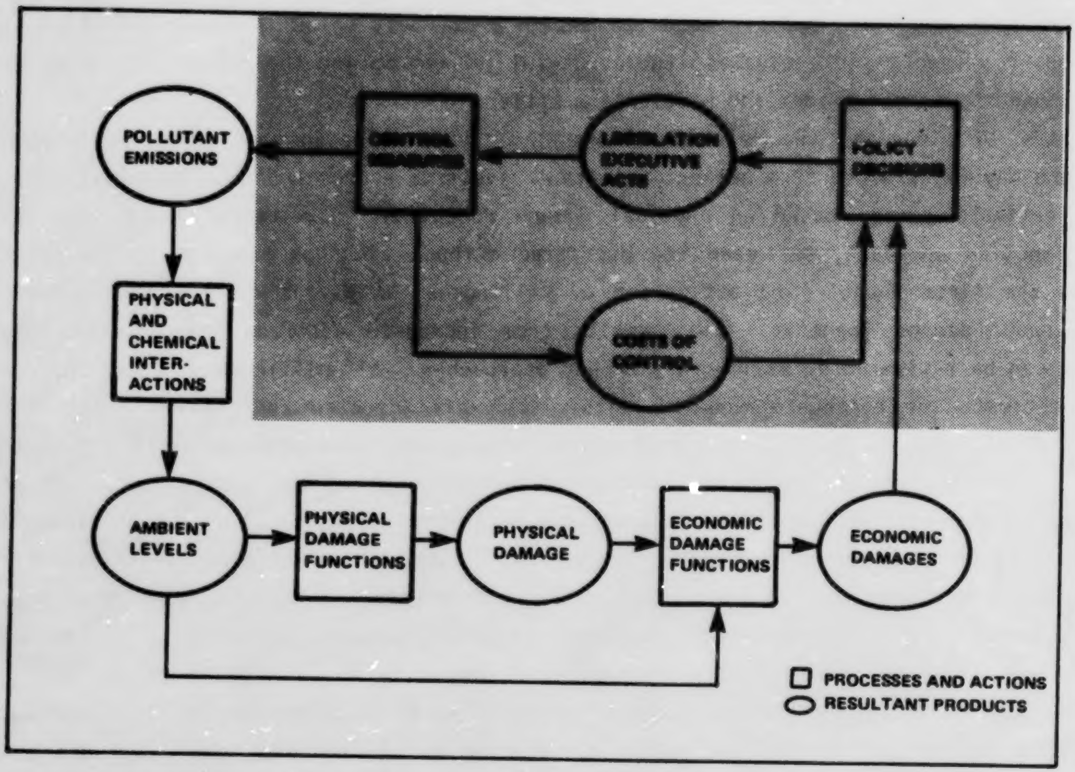


Figure 10-1. Relationship among emissions, air quality, damages and benefits, and policy decisions. Shaded area represents processes, actions, and resultant products outside the scope of this chapter.

Source: Hershaft (1976).

The physical damage function approaches are more extensively described here because they have been the most widely used. Therefore, it is important to explain briefly the method and some of its limitations. Nonmarket and indirect market studies are discussed in Section 10.5.

In the damage function approach, physical damage must be determined before economic damage can be estimated by calculating the exposure of certain materials in specific locations. Exposure is estimated from ambient air levels over specific time intervals. Depending on the pollutants and the kinds of material studied, both average short-term and long-term exposure data may be needed. These data are used to develop a physical damage function, which is a mathematical expression linking exposure to damage. The damage function is expressed in terms appropriate to the interaction of the pollutant and material. For example, the corrosion of metal might be expressed in units of thickness lost, while the deterioration of paint could be stated in units of reflectance or thickness lost. The pollutant level may be expressed in any concentration unit. Since material damage usually develops over time, average long-term pollutant levels are often used in calculating damage. Damage studies are often broken down by years, which helps economic comparisons and allows averaging of conditions in changeable climates.

A major problem in establishing reliable damage functions has been to separate the influence of the target pollutant from those of meteorological parameters and other air pollutants. For the corrosion of metals, RH is the most important variable. Rainfall, time of wetness, sunlight, and windspeed and direction are available for many locations and, if relevant, can be considered in damage calculations. The influence of atmospheric components should be considered in the context of the material being studied; for example, in studies of SO_x-related steel corrosion, even low atmospheric chloride levels may significantly affect the rate of damage.

A critical damage level should be determined before an economic value is placed on the incremental damage caused by pollution. This level represents the point at which the service life or functional utility of the material has ended or is severely impaired. When this point is reached, replacement or repair is necessary, and cost is incurred. For example, if a typical coat of paint is 60 μm thick, the critical damage level at which repainting is necessary may occur when about 10 μm remain. By comparing the amount of surface erosion in a clean environment with that in an area with a specific pollutant, the apparent damage from the pollutant can be calculated and used in assessing physical damage. The value is determined through economic damage functions, in which physical damage functions are coupled to the use and value of the material. Exposure, replacement, protection, and other data are included in the estimate. This approach cannot account for irreplaceable items, such as works of art, where the only measurable cost is that of preservation. The social cost of losses of historical artifacts cannot be quantified in monetary terms.

As discussed above in the context of Figure 10-1, most estimates of economic damage to materials have followed the damage function approach. This chapter first discusses the physical effects of SO_x and PM separately. Within these categories, laboratory studies are discussed first, then field studies. Results from field experiments are compared with laboratory-derived damage functions. When a damage function has been validated by field and laboratory experimental results, it may be used as a basis for determining total damage to materials caused by a given air pollutant.

In the damage function approach to cost estimation, physical damage is one component used in assessing economic damage. Other components used are a mathematical expression of surface area of materials exposed and cost factors associated with units of physical damage. Among the problems with this approach are:

1. Valid physical damage functions do not exist for all pollutants and materials.
2. Estimates of the amount and type of material exposed have usually been based on some surrogate such as production figures modified by service life data (field surveys of exposed material in place have not been reported).
3. Cost factors may or may not accurately reflect cost associated with pollution as opposed to other causes.

This latter consideration is especially a problem in estimates of soiling damage, where socioeconomic factors are heavily involved.

Past estimates of costs associated with PM, SO_x, or other pollutants must be considered in light of the above discussion. The last section of this chapter discusses cost estimates and their limitations for decisionmaking purposes.

10.2 SULFUR OXIDES

10.2.1 Corrosion of Exposed Metals

Sulfur oxides in the environment accelerate metal corrosion. Several factors other than concentration of SO_2 are important.

10.2.1.1 Physical and Chemical Considerations--The atmospheric corrosion of most metals is a diffusion-controlled electrochemical process. For electrochemical action to take place, the following are necessary: (1) an electromotive force between points on the metal surface; (2) a mechanism for charge transfer between the electronic conductors; and (3) a conduction path between the cathode and anode reaction centers. Measurements of the rate of SO_2-accelerated rusting of iron vary greatly from site to site, despite careful monitoring of pollutant concentrations, a fact that has often puzzled researchers. Several factors might be responsible for inconsistent results, including: (1) the deposition rate of gaseous or dissolved SO_2 and particles; (2) the variability in the electrochemical actions that cause corrosion; (3) the influence of rust on the rate of subsequent corrosion; (4) the interaction between the pollutant effects and "wetness time", often indicated by RH, on surface electrolyte concentrations; and (5) orientation of the metal surface.

The actual mechanism for the oxidation of SO_2 (and its hydrated products) at metal-water interfaces is little understood. Barton (1976) proposed the following schematic reaction:

$$SO_2 + O_2 + 2e \rightarrow SO_4^{2-} \qquad (10\text{-}1)$$

or

$$4HSO_3^- + 3O_2 + 4e \rightarrow 4SO_4^{2-} + 2H_2O \qquad (10\text{-}2)$$

The electrons are provided by the oxidation of the metal (M):

$$M \rightarrow M^{n+} + ne \qquad (10\text{-}3)$$

Duncan and Spedding (1974), using an electrophoretic method, found that the rates of SO_4^{2-} formation on iron and zinc surfaces were similar; the pseudo-first-order half-life (equivalent to corrosion product doubling time) was determined to be about 24 hours. Other workers (Karraker 1963; Yoshihara et al., 1964) reported higher oxidation rates (half-life, 10 to 100 minutes) in bulk solutions using Fe^{3+} catalysts, as summarized in Nriagu (1978).

In his discussion of long-term corrosion rates, Barton (1976) noted that rusting occurs first in localized areas and then spreads across the entire exposed surface. For iron, the formation of rust increases the adsorption of SO_2 dramatically depending on RH and temperature. For example, in an atmosphere of 10 ppm (26200 $\mu g/m^3$) SO_2 at >96 percent RH, virtually all the SO_2 that comes in contact with the corroded iron surface is adsorbed. On the other hand, corrosion products in zinc, copper, and, particularly, aluminum lessen the rate of SO_2 uptake. For the action of SO_2 on iron, Barton calculated that a surface uptake of 40 to 50 mg SO_2/m^2/day is required to accelerate corrosion, assuming critical RH and temperature. From this uptake level, he calculated that corrosion rate increase will be linearly dependent on SO_2 concentration. From the correlation between SO_4^{2-} rates and ambient SO_2 levels developed by Guttman (1968), Barton calculated that the surface uptake level of 40 to 50 mg SO_2/m^2/day corresponds to an average annual ambient air concentration of 0.015 ppm (40 $\mu g/m^3$) SO_2.

10.2.1.1.1 Relative humidity and corrosion rate. According to Schwarz (1972), the corrosion rate of a metal should increase by 20 percent for each increase of 1 percent in the RH above the critical RH value. It is evident that RH has a considerable influence on the corrosion rate, as established in laboratory trials by Haynie and Upham (1974) and Sydberger and Ericsson (1976). Although these experimental results do not support the exact rate predicted by Schwartz, it is apparent from Figures 10-2 and 10-3 (Haynie and Upham, 1974) that the corrosion rate of steel increases with increasing RH as well as with increasing SO_2 concentration.

The climate of an area is usually characterized by average RH rather than RH distribution. Since average RH is calculated from the distribution, there should be an empirical relationship between average RH and the fraction of time some "critical humidity value" (minimum concentration of water vapor required for corrosion to proceed) is exceeded, assuming

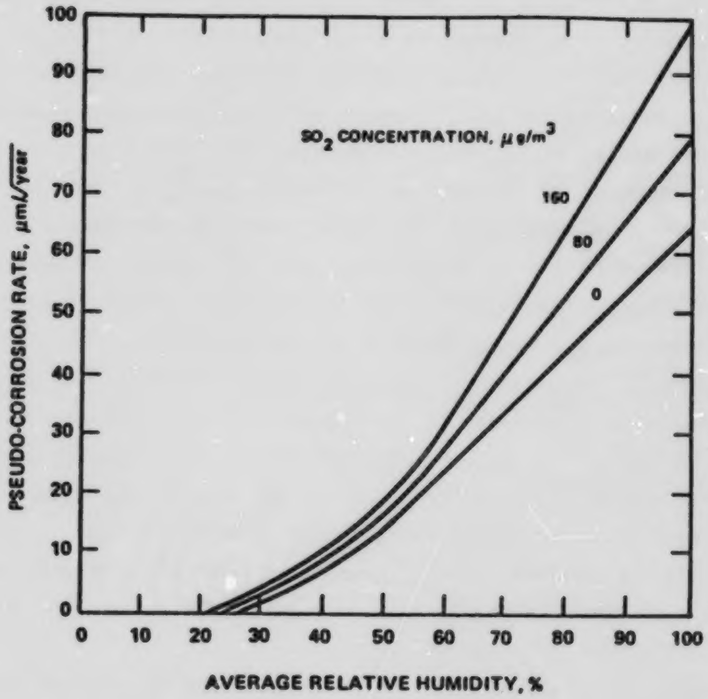


Figure 10-2. Steel corrosion behavior is shown as a function of average relative humidity at three average concentration levels of sulfur dioxide.

Source: Haynie and Upham (1974).

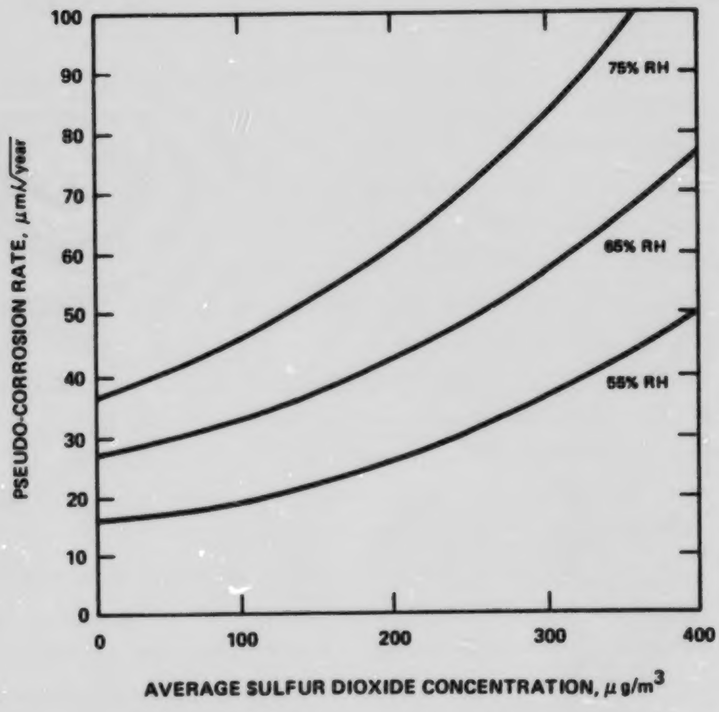


Figure 10-3. Steel corrosion behavior is shown as a function of average sulfur dioxide concentration and average relative humidity (RH).

Source: Haynie and Upham (1974).

a relatively constant standard deviation of RH (Mansfeld and Kenkel, 1976; Sereda, 1974). The fraction of time that the surface is wet must be zero when the average RH is zero and unity when the average RH is 100 percent. According to Haynie (1980), the following equation is the simplest single-constant first-order curve that can be fitted to observed data:

$$f = \frac{(1-k)\ RH}{100-k\ RH} \qquad (10\text{-}4)$$

where f = fraction of time RH exceeds the critical value

RH = average relative humidity

and k = an empirical constant less than unity.

Ten quarter-year periods of RH data from St. Louis International Airport were analyzed and fitted by the least-squares method to the above equation. The fraction of time the RH exceeded 90 percent gave a value of 0.86 for k. This fraction and the data points are plotted in Figure 10-4 (Haynie, 1980).

When the temperature of a metal is below the ambient dewpoint, water condenses on the metal surface. Whether or not the metal reaches the temperature at which condensation occurs varies with heat transfer between ground and metal and between air and metal. Condensation occurs when the RH adjacent to the surface exceeds a value in equilibrium with the vapor pressure of a saturated solution of whatever salts are on the surface. The solution may contain corrosion products, other hygroscopic contaminants, or both. Temperature, wind, sunshine, and night sky cover then become factors in establishing corrosion rates, since they determine whether there will be sufficient dew condensation.

Haynie (1980) reported on the relationship between diffusion theory and thermodynamics for the observed effects of five variables: pollution level, RH, temperature, wind velocity, and surface geometry. He observed that metals must be wet to corrode electrochemically. Surfaces are wet from condensation much more often than from precipitation. Further, while precipitation dilutes electrolytes, dew concentrates them.

10.2.1.1.2 Influence of rainfall on corrosion. Steel surfaces shielded from the leaching effect of rain may corrode at a higher rate than those exposed to rain. The SO_4^{2-} content of rust has been identified as a dominant factor in corrosion and is found at higher concentrations on surfaces sheltered from rain than on exposed surfaces because soluble SO_4^{2-} is leached from the rust. Sulfur deposition during rainfall, however, must also be considered. Haagenrud and Ottar (1975) noted that the rate of corrosion of unalloyed steel and zinc correlated with the amount of sulfur (SO_2 and SO_4^{2-}) in air and in precipitation.

As Kucera's (1976) review indicates, the mode of deposition complicates the analysis of the effects of acidic precipitation. For example, in areas where dry deposits of hydrogen and SO_4^{2-} ions exceed deposits in wet precipitation, flat steel plates corrode more rapidly on their

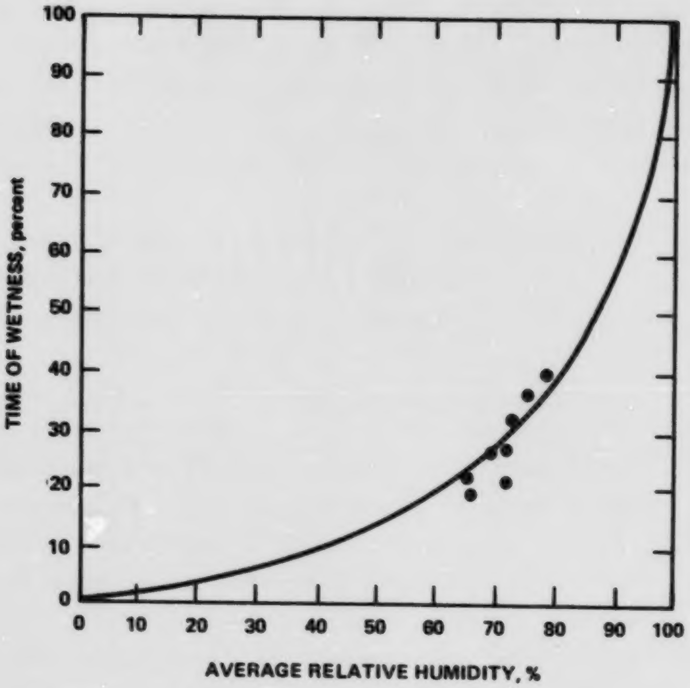


Figure 10-4. Empirical relationship between average relative humidity and fraction of time relative humidity exceeded 90 percent (time of wetness) is shown for data from St. Louis International Airport.

Source: Haynie (1980).

undersides than on their upper surfaces, suggesting that rainfall has more of a washing effect than a corrosive action (vide supra). In other areas, where wet and dry deposition are about equal, however, the upper sides of the plates corrode more quickly, suggesting that the corrosive effect of the rainfall predominates.

Matsushima et al. (1974), in studies of low-alloy weathering steels, considered the impact of the washing action of rain, the ease with which water would drain off the surface, and the drying effect of sunlight to determine the effect of these variables on the retention of particles that influence the electrolytic corrosion mechanism and the time of wetness. The authors hypothesized that the geometry of unpainted weathering steels may not favor the development of a protective oxide film of rust. The model structure used in the exposure trials contained horizontal and inclined roofs, vertical wall panels, and window frames.

Two sites were chosen: an industrial location and a residential site in the Kawasaki area, which has a cold, dry winter and a hot, humid summer. The results showed that the successful use of weathering steel is related not only to the severity of pollution but also to the specific interplay between shelter and the uniform washing action of rain. Thus, for areas in which the structural factors are unfavorable, the optimal rust film forms slowly and may deteriorate. Rust films develop and are then destroyed, and the surface never develops a protective film. Generally, boldly exposed surfaces such as horizontal or inclined roofs show the least corrosion.

Other variables, including amount and frequency of precipitation, and its pH level, humidity, and temperature, also determine the impact of acidic precipitation (Kucera, 1976). For a more thorough discussion of the role of acidic precipitation in corrosion, see Chapter 7.

10.2.1.1.3 Influence of temperature on corrosion. Guttman (1968) and Haynie and Upham (1974), using statistical techniques of multiple linear regression and nonlinear curve fitting, found no significant correlation between corrosion and temperature. Other studies, however, did find temperature to be a significant variable. Guttman and Sereda (1968) made continuous measurements of SO_x, time of wetness, and temperature in their outdoor exposure tests. The corrosion rate increased markedly with temperature. Barton (1976) found that the effect of increased temperature was more pronounced when the rust contained little water and SO_4^{2-}. Haynie et al. (1976) found that temperature is a significant variable in chamber studies of weathering steel.

The above results are in apparent conflict, but may be explained by Haynie's (1980) discussion of the effect of temperature on metal corrosion. He noted that if metal corrosion were activation energy-controlled, then the logarithm of the corrosion rate should be inversely proportional to the absolute temperature. In most cases, however, the rate of reaction is diffusion-controlled. Whether this control occurs in the environment or in the corrosion product film, the rate of corrosion is relatively insensitive to changes in ambient temperature. "A decrease in temperature," Haynie observed, "raises the relative humidity while it decreases diffusivity, thus normal temperature range effects on the overall corrosion rate will most likely not be observed. Freezing should produce a step decrease in the corrosion rate because diffusion is then through a solid rather than a liquid." (Haynie, 1980). This last supposition is supported by Biefer (1981) in his report of the first atmospheric corrosivity study of the Canadian Arctic and sub-Arctic. Lowest average rates of penetration (2 to 5 µm/yr) were recorded at mainland sites removed from the seacoast; the highest rates (21 to 34 µm/yr) were those recorded at sites less than 1 km from the sea. Biefer (1981) attributed the higher corrosion rates to "localized factors such as atmospheric sulfur dioxide, atmospheric chlorides, and factors relating to the time-of-wetness of the corroding surface." Sereda (1974) found that at -20°C metal corrosion is slowed, but not halted.

10.2.1.1.4 Hygroscopicity of metal sulfates. The SO_4^{2-} in rust stimulates further corrosion by a mechanism related to the critical RH at which an electrolyte film is formed. The hygroscopicity of iron sulfates in the rust lowers the critical RH for corrosion; however, sulfates are not the most deliquescent salts. For example, chloride and nitrate salts, which have higher hygroscopicity than sulfates, make corrosion possible at lower humidities.

Surfaces contaminated by sea salt (mostly NaCl) can be expected to be wet when the RH exceeds 75 percent. In contrast, calcium chloride keeps surfaces wet at an RH as low as 30 percent. A saturated solution of zinc sulfate at 20°C is in equilibrium at 90 percent RH. Thus, zinc corroded by SO_2 is expected to be wet when the RH exceeds 90 percent (Haynie, 1980).

10.2.1.1.5 Electrical conductivity of rust. Barton (1976) postulated that SO_4^{2-} ions influence the anodic dissolution of iron as a function of their concentration at the steel-rust interface. The corrosion rate of the rust layer is based in part on the high electronic conductivity of rust, which allows the reduction of oxygen within the rust layer. The rate is also influenced by the porosity of rust, which permits rapid diffusion of oxygen to the cathode.

In the presence of SO_2, $FeSO_4$ is formed before insoluble rust develops. The amount of SO_2 required is small; each SO_2 molecule can generate 20 to 30 molecules of rust. Once $FeSO_4$ is formed, rusting can continue even though SO_2 is no longer present in gaseous form.

10.2.1.1.6 Electrical reduction of rust. Evans (1972) suggests that oxidative hydrolysis of $FeSO_4$ occurs slowly, and would be important only in the initial stage of corrosion. He proposes that there is a rate-controlling cathodic process. Thus, the corrosion products in the ferric state would be converted to magnetite (Fe_3O_4) by a reaction involving the reduction of ferric oxyhydroxide (FeOOH):

$$Fe^{2+} + 8FeOOH + 2e^- \rightarrow 3Fe_3O_4 + 4H_2O \qquad (10\text{-}5)$$

10.2.1.1.7 Corrosion-protective properties of sulfate in rust. The rust layer on steel is somewhat protective against further corrosion, though far less so than the corrosion layer on zinc and copper. The content of soluble compounds in rust limits its protection of steel.

Rust samples investigated by Chandler and Kilcullen (1968) and Stanners (1970) contained 2 to 2.5 percent soluble SO_4^{2-} and 3 to 6 percent total SO_4^{2-}. The outer rust layer contained a small amount (0.04 to 0.2 percent) of soluble SO_4^{2-}, compared with 2 percent in the inner rust layer. The concentration of insoluble SO_4^{2-} was fairly uniform throughout the rust layers.

The emphasis on the composition of the rust layer has led to studies of the corrosion-protective properties of rust as a function of exposure history (Nriagu, 1978; Sydberger, 1976). Steel samples initially exposed to low concentrations of SO_x and then moved to sites of higher SO_x concentrations corroded at a slower rate than did samples continuously exposed to the higher concentrations. Exposure tests started in summer showed slower corrosion rates during the first years of exposure than those started in winter.

The long-term corrosion rate of steel appears to depend on changes in the composition and structure of the rust layer. During the initiation period, which varies with the SO_2 concentration and other accelerating factors, the rate of corrosion increases with time (Barton, 1976). Because it is porous and nonadherent, the rust initially formed offers no protection; in fact, it may accelerate corrosion by retaining hygroscopic sulfates and chlorides, thus producing a microenvironment with a high moisture content. After the initiation stage, the corrosion rate decreases as the protective properties of the rust layer improve. Satake and Moroishi (1974) relate this slowing down to a decrease in the porosity of the rust layer. During a third and final stage, corrosion attains a constant rate and the amount of SO_4^{2-} in rust is proportional to atmospheric SO_x concentrations.

The quantitative determination and subsequent interpretation of corrosion rates becomes difficult if it is not known how long the metal has had a surface layer of electrolyte. Variations in the "wet states" occur with RH, temperature, rain, dew, fog, evaporation, wind, and surface orientation. The surface electrolyte layer may form on a metal surface as a result of rain, dew, or adsorption of water from the atmosphere. Capillary condensation in rust can be related to the minimum atmospheric moisture content that allows corrosion to occur (i.e., critical RH). Centers of capillary condensation of moisture on metals can occur in cracks, on dust particles on the metal surface, and in the pores of the rust (Tomashov 1966).

10.2.1.2 Effects of Sulfur Oxide Concentrations on the Corrosion of Exposed Metals--Most of the laboratory studies reviewed in this section have measured corrosion rates related to exposure to SO_2 alone or in combination with other compounds. In field exposure studies, where SO_x almost invariably occur in combination with other airborne pollutants, an attempt is made to assign separate values to SO_x and to describe pollutant interactive effects on corrosion. The discussion here unavoidably overlaps somewhat with a later section on particles, since SO_2 contributes to formation of secondary sulfates. Here, the emphasis is on the direct role of SO_x in the corrosion process (e.g., the oxidation of SO_2 with moisture on a metal surface). In a later section, sulfates are discussed in terms of their indirect role (i.e., their ability to increase wetness time of a metal surface).

10.2.1.2.1 Ferrous metals. Ferrous metal products and structures are exposed widely to ambient pollutant levels. Rusting of these metals is the best documented form of metallic corrosion affected by SO_x. This subsection reviews studies of rusting rates of ferrous metals, such as iron, steel, and steel alloys.

A number of investigators reported data from 1959-1968 that showed that the addition of 0.05 to 0.5 percent copper to steel results in improved corrosion resistance (Larrabee, 1959; Larrabee and Coburn, 1962; Brauns and Kalla, 1965; Schwenk and Ternes, 1968; and Barton, 1976).

Stainless steels contain more than 12 percent chromium and are widely used in outdoor exposures. They are specified for use in many industrial processes involving corrosive liquids that rapidly attack ordinary steels. The high corrosion resistance of stainless steels that incorporate chromium, molybdenum, and nickel is attributed to the protective properties of the oxide film formed on these alloys. In heavily polluted atmospheres, however, this film is not completely protective. Particles in settled dust, including sulfates and chlorides, can promote rupture of the oxide film and cause pitting corrosion, which may be influenced by the surface finish (see Section 10.3.1).

The lowest alloyed stainless steels have little corrosion resistance. In particular, #13 Cr steel suffers pitting attack in industrial atmospheres. Ergang and Rockel (1975) report that the austenitic steels of 18-percent Cr and 8-percent Ni are reasonably resistant in urban atmospheres but have shown slight rusting in industrial areas. The rusting rate is decreased when the steel surface is cleaned of atmospheric deposits.

10.2.1.2.2 Laboratory and field studies emphasizing ferrous metals. It is useful to consider laboratory and field studies of corrosion effects separately because cause and effect is much clearer in laboratory experiments; field studies are often beset by confounding variables. Laboratory studies develop from controlled experiments, including as many variables as are thought likely to influence damage to the material being studied. Data on materials damage result from exposure to various concentrations of the air pollutant being studied. Analysis of the data resulting from such laboratory studies is used to develop a mathematical expression of the relationship between the concentration of a pollutant and damage to materials. Such a mathematical expression is called a damage function; that is, the quantitative expression of a relationship between exposure to specific pollutants and the type and extent of damage to a target population. Factors that are shown to be significant influences in producing damage are included in the damage function.

Information derived from laboratory studies is used to design field experiments, which are performed to test the possibility of extrapolating laboratory results to ambient conditions. The parameters that are measured in field studies are those found or suspected to be important factors in the laboratory experiments. The results of laboratory experiments do not easily translate to field situations, however, since ambient air pollution levels and other influencing environmental factors vary widely both in time and space. Temporal variables include fluctuations in temperature, wind moisture content, insulation, rainfall, and its chemical characteristics (e.g., acidic rain), and atmospheric pollutant concentrations. Spatial factors include differences such as aspect, altitude, electromagnetic fields, and indigenous microorganisms. Initial conditions of the material being studied must also be considered.

The results of field studies are compared with the laboratory-derived damage function. In some cases the results are comparable, and the laboratory damage function is validated. In other cases the data analysis may result in a markedly different damage function, with more or fewer variables.

10.2.1.2.2.1 Laboratory studies. Spence and Haynie (1974) described the design of a laboratory experiment to identify the effects of environmental pollutants on various materials including ferrous metals. The environmental system consisted of five exposure chambers to control temperature and humidity, and chill racks to simulate the formation of dew. Gaseous pollutants included those usually monitored in field exposures: SO_2, NO_2, and O_3. Experiments were statistically designed for analysis of variance, and a system was selected to study the interactive effects of pollutants and other variables. The effect of particles was not included in the design. The chambers were equipped with a xenon arc light to simulate sunlight. The system was designed to maintain air contact with the various materials at preselected temperatures, RH's, flowrates, and pollutant concentrations. A dew-light cycle was used; it produced faster deterioration than did conditions of constant humidity and temperature.

Haynie et al. (1976 and 1978) exposed weathering steel in the chamber study described above and measured concentrations of SO_2, NO_2, and O_3 in various combinations and at two levels of pollutant concentration. Ozone was of interest since an earlier field experiment (see Section 10.2.1.2.2, Field Studies) had indicated that the presence of oxidants inhibited metallic corrosion. The corrosion rate was measured by loss in weight of the weathering steel. Six panels each were exposed to 16 polluted-air and 4 clean-air conditions; measurements were taken at 250, 500, and 1000 hours of exposure. The weight losses were converted to equivalent thickness loss values. As expected, corrosion was most severe at high SO_2 concentrations and high humidity. Ozone neither inhibited nor accelerated corrosion. The authors concluded that some other oxidant or unmeasured factor that was covariant with O_3 caused the inhibition effect. If the data from the sites with high oxidant concentrations in the field exposure experiments were excluded, however, the damage from the laboratory study was an excellent predictor of the field results. The coefficient of determination for the field data using the laboratory function was 0.986. The following physical damage functions were developed by Haynie et al. (1976) to relate SO_2 concentrations to weathering steel panels and galvanized steel corrosion, respectively ($R^2 = 0.91$).

$$\text{corrosion} = [5.64\sqrt{SO_2} + e^{(55.44 - 31{,}150/RT)}]\sqrt{f_w} \qquad (10\text{-}6)$$

$$\text{corrosion} = (0.0187\ SO_2 + e^{41.85 - 23{,}240/RT})\ t_w \qquad (10\text{-}7)$$

where:

corrosion is expressed in μm thickness lost

SO_2 = sulfide dioxide concentration expressed in $\mu g/m^3$

f_w = fractional time of panel wetness

t_w = time of wetness in years

R = the gas constant (1.9872 cal/gm mol/K)

T = geometric mean temperature of panels when wet, K.

Sydberger and Ericsson (1976) studied the corrosion of mild steel at 1, 10, and 100 ppm (2620, 26,200, and 262,000 $\mu g/m^3$) SO_2 across the range of critical humidities (80 to 96 percent RH). The flowrate of the SO_2 atmosphere was varied, and some samples were sprayed with water to simulate rain or condensation. The chemical composition of the corrosion products was studied by X-ray diffraction, infrared spectrometry, and electron spectroscopy for chemical analyses (ESCA) techniques. The flowrates of the SO_2 atmospheres markedly influenced the corrosion rates. It appears that corrosion rates are related not only to the SO_2 concentration in the atmosphere, but also to the supply of SO_2 per unit surface area and time. Spraying the samples with distilled water at intervals substantially increased corrosion.

Sydberger and Ericsson's (1976) based their analysis of the corrosion product (rust layer) on the concepts of Schwarz (1972) and Barton (1976) that SO_4^{2-} is the primary corrosion stimulant in rust formation. Anodic activity is maintained by the concentration of ferrous sulfate in the electrolyte. An SO_2 supply of 4 $\mu g/cm^2/hr$ at the lowest humidity initiated corrosion at a low rate. A rise above 50-percent RH increased corrosion markedly. Of particular interest was the finding that different flow rates at 1 ppm (2620 $\mu g/m^3$) SO_2 with 96-percent RH gave significantly different corrosion rates. This study of the effect of rust on corrosion showed that even at high humidity and high SO_4^{2-} content, the corrosion rate decreased to a low level when the SO_2 concentration was low.

10.2.1.2.2.2 Field studies. For outdoor exposures, eddy diffusion is the primary rate-controlling factor in the delivery of pollutants to a surface. This flux is not constant and is a function of the horizontal wind velocity gradient away from the surface. The transport of a pollutant to a surface is usually expressed as a "deposition velocity", defined as the flux to the surface divided by the ambient pollution level at some specific measuring height.

Reported deposition velocities for gaseous pollutants have usually been within an order of magnitude of 1 cm/sec. These values are consistent with calculated estimates based on an analogy with momentum flux and measured wind velocity profiles (Sydberger, 1976).

The amount of SO_2 reaching a steel surface depends on wind direction, wind velocity, and the orientation of the surface to the emission source. The concept that SO_2 deposition varies with flow direction and velocity suggests that data on concentration alone cannot be used to determine the supply of SO_2 to metal surfaces; therefore surface adsorption methods like the lead candle method provide valuable information in relating supply of SO_x to metal surfaces (Sydberger, 1976). Upham's (1967) work indicated, however, that corrosion of mild steel at seven Chicago sites increased with time and with increasing mean SO_2 concentration (Figure 10-5). Difficulties in relating sulfation measurements to ambient SO_2 measurements are discussed in Chapter 3.

Haynie and Upham (1971) continuously monitored urban pollutants including SO_2, NO_2, and O_3 (oxidants) to determine whether previously unconsidered variables might affect steel corrosion. They also considered temperature, RH, and TSP. Their 5-year program, begun in 1963, involved sites in Chicago, Cincinnati, New Orleans, Philadelphia, San Francisco, Washington, Detroit, and Los Angeles. They studied three types of steel expected to show different levels of resistance to atmospheric corrosion: (1) a plain carbon steel containing some copper (0.1 percent copper); (2) a copper-bearing steel (0.22 percent copper); and (3) a low-alloy weathering steel (0.4 percent copper with 0.058-percent phosphorus). The exposure periods were 4, 8, 16, and 32 months. The same steels were exposed at rural sites as a control. The rural sites proved to have higher-than-expected corrosion rates; however, SO_2 concentrations were not measured at these sites. Multiple regression analysis established significant correlations between average SO_2 concentrations and corrosion of all three steels at the urban sites. Temperature was not a statistically significant variable. Average RH was insignificant because the range was only between 62 and 77 percent.

Inspection of the monitored SO_2 and oxidant concentrations revealed wide variations from site to site. Multiple-regression analysis showed that high concentrations of oxidants correlated with lowered metallic corrosion rates. The resulting physical damage functions included terms for SO_2, oxidants, and time of exposure. A more recent laboratory investigation by Haynie et al. (1976), however, (see preceding section) has shown that O_3 is not a significant corrosion controlling variable.

Mansfeld (1980) made observations at nine test sites in and around St. Louis for 30 months beginning in October 1974 as part of the Environmental Protection Agency's Regional Air Pollution Study to determine the effect of airborne pollutants on galvanized steel, weathering steel, stressed aluminum, marble, and house paint--essentially the same materials examined in the chamber study reported by Haynie et al. (1976). During 1975 and 1976, atmospheric corrosion monitors (ACM) of the type described by Mansfeld and Kenkel (1976) were installed at

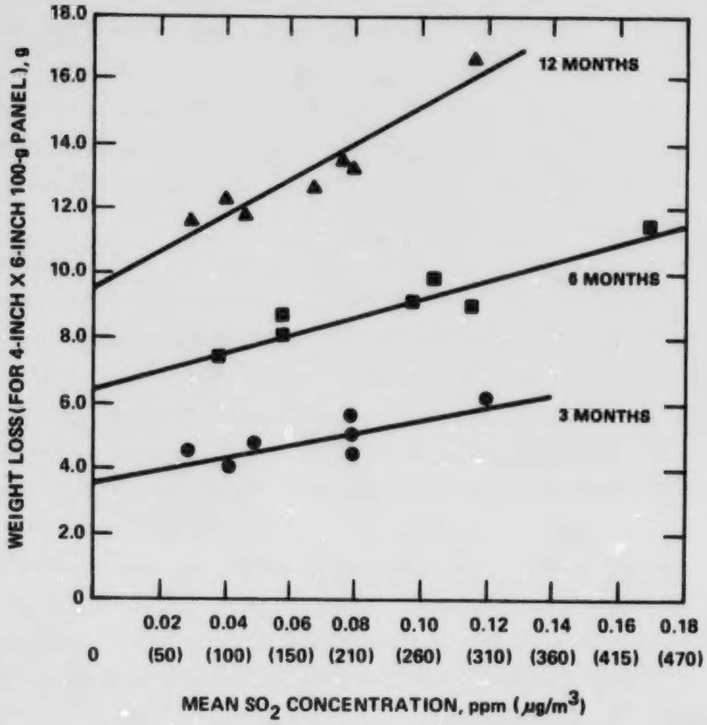


Figure 10-5. Relationship between corrosion of mild steel and corresponding mean SO_2 concentration is shown for seven Chicago sites. (Corrosion is expressed as weight loss of panel.)

Source: Upham (1967).

four sites to measure time of wetness. Each ACM consists of a copper-zinc or copper-steel couple that registers current flow when an electrolytic path forms between the two plates as a result of deposition of water from the air, dew, or rain on corrosion products. The ACM measures the time that the panel is wet enough for the electrochemical mechanism of corrosion to occur. Parameters measured in Mansfeld's study included windspeed, wind direction, temperature, O_3 concentration, total hydrocarbon concentration, total sulfur and NO_x concentration, hydrogen sulfide concentration, SO_2 concentration, RH, SO_4^{2-} concentration, NO_3^- concentration, TSP concentration, and time of wetness.

Mansfeld determined weight losses for galvanized steel, weathering steel, house paint, and marble (he removed aluminum tension samples after failure). His data show that damage to a particular material does not necessarily occur at the same corrosion rate at each site. Preliminary statistical analysis of the results failed to show significant correlation between corrosivity and pollutant concentrations. There was substantial error in the measurement of RH, an extremely important corrosion variable. Relative humidity was, therefore, not included in the regression analyses. Concentrations of SO_2 measured by Mansfeld (1980) were generally an order of magnitude lower than the 130 ppb (340 $\mu g/m^3$) concentrations reported by Upham (1967) at urban sites in St. Louis.

In another study Haynie and Upham (1974) exposed enameling steel containing 0.019-percent carbon and 0.028-percent copper at 57 sites in National Air Surveillance Network (NASN). They measured corrosion by weight loss and quantitatively determined other pollutants, including gaseous SO_2, TSP, and the amount of SO_4^{2-} and NO_3^- in the particles. (For a discussion of the measurement methodology for NASN, see Chapter 3.) Sulfur dioxide levels ranged from 9 to 374 $\mu g/m^3$, TSP from 11 to 182 $\mu g/m^3$, and RH from 29 to 76 percent. The average temperature remained within a fairly narrow range and was considered constant. The temperature, the quantity of total particles, and the presence of NO_3^- in the particles did not significantly affect the corrosion rate of steel. The concentration of SO_2 was significant only when SO_4^{2-} was not included in the regression analysis. At each site, SO_4^{2-} content of the particles and SO_2 concentration were closely related. On the basis of this study, Haynie and Upham derived the following empirical expression to obtain the best relationship between corrosion of enameling steel and atmospheric SO_2 content:

$$\text{Corrosion} = 325\sqrt{t}\, e^{(0.00275\ SO_2\ -\ 163.2/RH)} \qquad (10\text{-}8)$$

where: corrosion is expressed as μm

SO_2 = sulfur dioxide expressed as $\mu g/m^3$

RH = average relative humidity (percent)

t = time in years

Average RH is a substitute variable for the fraction of time the steel is wet.

Considerable effort has gone into isolating environmental variables that predict long-term corrosion rates. Empirical expressions for corrosion of various steels exposed to the atmosphere (see Table 10-1) have been developed by Chandler and Kilcullen (1968), and Haynie and Upham (1974). These equations may be used to relate reduction in SO_2 and sulfates to reduction in corrosion of metals, serving as a basis for a benefit appraisal.

All of the above-discussed damage functions are statistical models developed from regression analysis of the correlations of air chemistry and effects data resulting from laboratory or field exposure experiments. Such functions vary in form, reflecting differences in parameters measured and/or measurement methodology. For instance, if the function includes a term for SO_2 expressed as $\mu g/cm^2/day$, then deposition velocity is already accounted for. If, alternatively, SO_2 is expressed as $\mu g/m^3$, then deposition velocity may be accounted for in the equation. Barton (1976) proposed an equation for metal corrosion rate which combines both kinetics of corrosion processes specific to the metal of interest and results of regression analysis. Such an equation may be more universally applicable, although the statistical damage functions discussed previously should be applicable across a range of environmental conditions normally encountered in the United States. Barton's proposed equation is:

$$\upsilon_K = M \cdot \tau^n \cdot S^m \qquad (10\text{-}9)$$

where

υ_K = corrosion rate of metal in μm/yr

M = constant involving specific corrosion kinetics of the metal

τ = time of wetness, long-term average, expressed as hrs/day with RH $\geq$ 80% and temperature $\geq$ 0°C

S = SO_2 expressed as $mg/m^2/day$

n and m = constants determined by regression analysis

10.2.1.2.3 Comparison of ferrous and nonferrous metals. Sydberger and Vannerberg (1972) examined the influence of RH and rust on the adsorption of SO_2 on metal surfaces, using radioactive sulfur. The concentration of SO_2 was 0.1 ppm (262 $\mu g/m^3$), and RH varied between 50 and 98 percent. Polished and preexposed samples of iron, zinc, copper, and aluminum were compared for their adsorption properties. Iron, zinc, and copper were preexposed to SO_2 concentrations (100 ppm or 2.6×10^5 $\mu g/m^3$) at 98 percent RH and 22°C for 3 hours. The aluminum samples were preexposed for 30 hours. The principal corrosion product identified by X-ray diffractometry was hydrated metal sulfate. Adsorbed SO_2 was measured at 30-minute intervals with a Geiger counter. The corrosion rate at 90-percent RH was initially high for zinc and copper but quite low for aluminum. Adsorption of SO_2 on preexposed iron samples was high. At 80-percent RH, almost all of the SO_2 was adsorbed. The high adsorption rate is perhaps explained by the rapid oxidation of adsorbed SO_2 caused by the catalytic effect of the rust.

TABLE 10-1. SOME EMPIRICAL EXPRESSIONS FOR CORROSION OF EXPOSED FERROALLOYS

Exposure Material	Site	Study Duration	Empirical Equation	Parameter Units	Source	Note/Comments
Steel A*	Sheffield England	1 year	$y = 0.51 + 0.01x$	y = corrosion rate in mils/year $x = SO_2$ in μg/m^3	Chandler and Kilcullen (1968)	Authors stated that SO_2 and smoke has a major influence on the corrosion rate of steel and accounted for about 50% of the variations found at the different sites. Other factors, such as time of wetness, were found equally important in determining the corrosion rate of steel. 95% confidence limit ± 0.75 mils for any point on regression.
Steel B*	Sheffield England	1 year	$y = 0.82 + 0.006x$	y = corrosion rate in mils/year $x = SO_2$ in μg/m^3	Chandler and Kilcullen (1968)	95% confidence limit ± 0.5 mils for any point in regression.
Enameling steel	NASN sites	1-2 years	$corr = 183.5 \sqrt{t}\, e^{[0.06421 sul - (163.2/RH)]}$	corr = depth of corrosion, μm sul = average level of sulfate in suspended particulate, μg/m^3 RH = average relative humidity, percent t = time in years	Haynie and Upman (1974)	std. dev. 85.0; 95% conf. limits 13.4 lower, 353 upper std. dev. 0.00451; 95% conf. limits 0.0552 lower, 0.0732 upper std. dev. 32.8; 95% conf. limits 97.6
Enameling steel	NASN sites	1-2 years	$corr = 325 \sqrt{t}\, e^{[0.00275\, SO_2 - (163.2/RH)]}$	parameter units same as for previous equation except SO_2 = μg/m^3 SO_2	Haynie and Upham (1974)	Derived by substitution from above equation, based on sufate and SO_2 relationship defined as: $sul = 8.9 + 0.0429\, SO_2$, std. deviation ± 8.2.

Note: 1 ppm SO_2 = 2620 ug/m^3.

* composition of steels A and B are as follows:

Element	% Steel A	% Steel B
carbon	0.058	0.20
silicon	trace	0.04
sulfur	0.024	0.052
phosphorous	0.013	0.038
manganese	0.335	0.58
tin	0.015	-
copper	0.060	0.28
nickel	0.040	0.10
aluminum	0.015	-

The initial rate of adsorption on polished iron below 80 percent RH is related to the absence of corrosion; however, at increased humidity corrosion is initiated and the adsorption rate increases.

The observation that SO_2 adsorption can take place at humidities below the critical humidity (Sydberger and Vannerberg, 1972) is of particular significance. This suggests that SO_2 will be adsorbed on a rusty iron surface during periods of low humidity and will affect the corrosion rate when humidity rises. Table 10-2 shows the critical humidities for non-ferrous metallic surfaces, as summarized from Nriagu (1978) and National Research Council (NRC, 1979). The corrosion products of copper and aluminum have an extremely low adsorption capacity below 90-percent RH, confirming the lower sensitivity of these metals to corrosion by SO_2 (see Figure 10-6, from Sydberger and Vannerberg, 1972).

TABLE 10-2. CRITICAL HUMIDITIES FOR VARIOUS METALS

Metal	Critical humidity, percent RH
Aluminum	75-80
Brass	60-65
Copper	65-70
Nickel	65-70
Zinc	70-75

Aluminum is generally considered to be corrosion resistant. It is quite resistant to SO_x when RH is less than 50 percent. At higher humidities aluminum can corrode rapidly in the presence of very high SO_2 concentrations forming a hydrated aluminum sulfate $[Al_2(SO_4)_3 \cdot 18H_2O]$ surface deposit. At low concentrations of acid sulfate particles it forms a protective film of aluminum oxide. When the film becomes contaminated with dirt and soot particles, however, there is a change in surface appearance characterized by mottling and pitting. Simpson and Horrobin (1970) reported that aluminum undergoing long exposure in industrial areas displayed white areas of crystalline corrosion products. Aluminum surfaces exposed for more than 5 years to an SO_4^{2-} concentration of 550 $\mu g/m^3$ had pits as deep as 14 mils (0.36 mm). This is, of course, some 50 times higher than typical ambient concentrations (see Chapter 5).

Fink et al. (1971) summarized measured corrosion rates and depth of pitting of aluminum surfaces in rural, mild industrial (30 $\mu g/m^3$ or 0.01 ppm SO_2), normal industrial (370 $\mu g/m^3$ or 0.14 ppm SO_2 and 80-percent RH), and severe industrial areas. Their overall conclusion was that, although some loss of thickness occurred in the first 2 years, structures composed of aluminum and its alloys are resistant to air pollutants.

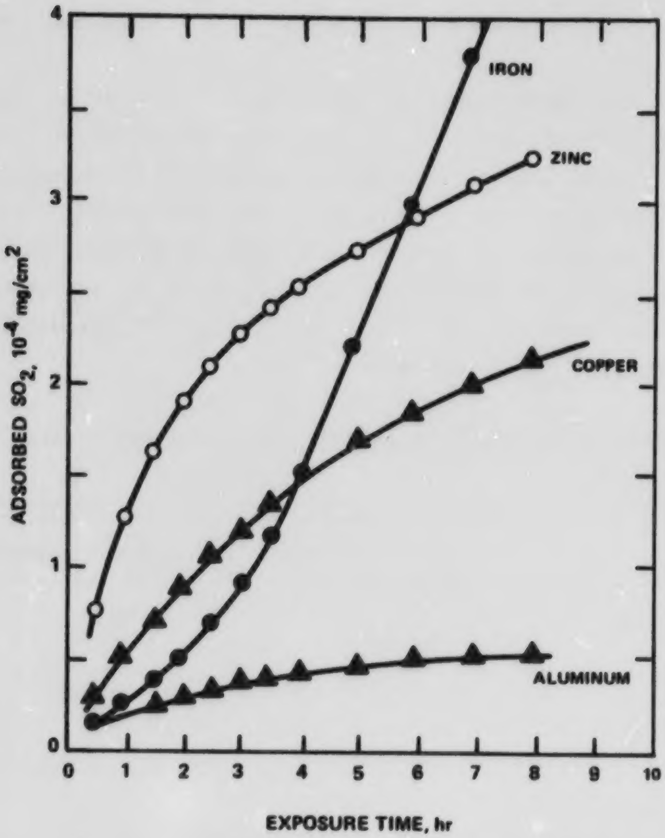


Figure 10-6. Adsorption of sulfur dioxide on polished metal surfaces is shown at 90 percent relative humidity.

Source: Adapted from Sydberger and Vannerberg (1972).

In another study, Haynie (1976) found that stressed aluminum specimens exposed to SO_2 at concentrations of 79 (0.03 ppm) and 1310 μg/m^3 (0.5 ppm) lost approximately 8.6 and 27.6 percent of their bending strength, respectively. He also noted that 7005-T53 high-strength aluminum alloy tubing, which contains very little copper, is susceptible to stress-corrosion cracking in industrial environments.

Abe et al. (1971) exposed copper and copper alloys for 2 years in marine, rural, highly industrial, and urban areas in which there was great variation in pollutant and salt content. Analyses of the surface deposits showed the presence of basic sulfate, sulfide, and chloride of copper. The green patina on copper was analyzed and determined to be basic copper sulfate in urban areas and basic copper chloride in seacoast areas. These surfaces were protective against further corrosion.

According to Simpson and Horrobin (1970), the formation of these basic copper salts can take as long as 5 or more years and will vary with the concentration of SO_4^{2-} or chloride particles, the humidity, and the temperature. They reported the corrosion rate of copper to be 0.9 to 2.2 µm/year in industrial atmospheres, compared with 0.1 to 0.6 µm/year in rural areas.

Sydberger (1976) attributed the high corrosion resistance of nickel and copper compared with unalloyed steel to the ability of these metals to form a layer of insoluble basic SO_4^{2-} that protects the metal surface. Such layer formation does not occur on steel.

10.2.2 Protective Coatings

Susceptible materials are generally coated for protection against the effects of exposure. The coatings provide either sacrificial protection or barrier protection. In galvanization, zinc is applied to ferrous metal for sacrificial protection. Thus, while the galvanized surface may suffer corrosion damage, it helps to prevent rusting of steel products such as gutters, cables, wire fencing, and building accessories. Barrier protection is provided by varnishes, lacquers, and paints by sealing the underlying surface material against intrusion by moisture.

10.2.2.1 Zinc-Coated Materials--Zinc is generally exposed as a protective coating for steel products since zinc coating is fairly resistant to atmospheric corrosion. Zinc is anodic with respect to steel; when zinc and steel are in contact with an electrolyte, the electrolytic cell provides current to protect the steel from corrosion with some oxidation of the zinc.

Guttman (1968) carried out a long-term exposure of zinc panels with measurement of the atmospheric factors. He found that zinc is corroded by SO_2 and that time of wetness and concentration of SO_2 are the major factors that determine the rate of corrosion.

Fleetwood (1975) conducted 5-year exposure studies of zinc and iron in a number of locations ranging from dry tropical to industrial. He estimated the service life of galvanized steel to be 15 to 20 years in an industrial area containing pollutants and 300 years in a dry tropical unpolluted area. Kucera (1976) noted strong correlations between the corrosion rate and (1) the adsorption of SO_2 on zinc surfaces and (2) the concentrations of SO_2.

Haynie and Upham (1970) exposed zinc panels in eight cities, continuously monitoring SO_2 concentration, and collecting meteorological data, including temperature and RH, from the nearest weather stations. They developed the following empirical equation, which correlates corrosion rate with average SO_2 concentration (for the study range of 10 to 479 µg/m^3) and RH:

$$Y = 0.001028\,(RH - 48.8)\,SO_2, \qquad (10\text{-}10)$$

where

Y = zinc corrosion rate (µm/yr),
RH = average annual RH in percent, and
SO_2 = average SO_2 concentration (µg/m^3).

The regression intercept indicated that no corrosion would occur below an average RH of 48.8 percent. This expression gave a reasonably good linear fit with the experimental corrosion results obtained by Haynie and Upham for SO_2 concentration and RH.

Haynie (1980) performed a regression on the products of the coefficients of the studies listed in Table 10-3 and SO_2 levels. The results of the regressions yielded 1.087 with a standard deviation of ± 0.283. Based on this analysis and the St Louis study results, Haynie (1980) expressed the relationship of the corrosion of small specimens of galvanized steel as follows:

$$C_z = 2.32\, t_w + 0.0134\, v^{0.781} \cdot SO_2 \cdot t_w, \qquad (10\text{-}11)$$

C_z = corrosion in μm

t_w = time of wetness in years,

v = wind velocity in m/s

SO_2 = μg/m^3

Because the equation was derived from the results of regression for several studies, not an original data set, no R^2 was calculated. A theoretical damage coefficient (for present purposes, damage = any measurable adverse effect) for a pollutant can be calculated from the stoichiometry of a reaction and the deposition velocity. For the reaction between SO_2 and zinc to form zinc sulfate, the coefficient is 0.045, when the zinc corrosion rate is expressed in micrometers per year, SO_2 in micrograms per cubic meter, and the deposition velocity (u) in centimeters per second. For a small zinc or galvanized steel sheet specimen, the material damage coefficient for SO_2 is calculated to be $0.0123\, v^{0.78}$, which agrees well with the above determined empirical coefficient. At a wind velocity of 4 m/sec, the value is 0.0363 (μm/yr)/(μg/m^3). For the same conditions, a similar calculation for marble yields a coefficient of 0.136 (μm/yr)/(μg/m^3).

Haynie (1980) restudied the results of six exposure investigations to relate the corrosion of zinc and galvanized steel to the concentration of SO_2. Each investigation was different and the data were evaluated differently; thus, no direct comparison of the results as they were published was possible. In reevaluating the data from each study, however, Haynie used techniques that permitted the comparison of the various data sets.

Table 10-3 compares the experimental regression coefficients obtained from all of these studies. The SO_2 coefficient for the chamber study is low, whereas the analogous coefficients for the Community Air Monitoring Program (CAMP) (Haynie and Upham, 1970) and Interstate Surveillance Program (ISP, Cavender et al., 1971) studies are high and agree with each other. The remaining three SO_2 coefficients are generally in good agreement. The time-of-wetness coefficients are all within a range of ± 0.75 from a mean of 1.73 μm/yr.

TABLE 10-3. EXPERIMENTAL REGRESSION COEFFICIENTS WITH ESTIMATED STANDARD DEVIATIONS FOR SMALL ZINC AND GALVANIZED STEEL SPECIMENS OBTAINED FROM SIX EXPOSURE SITES

Study	Time-of-wetness coefficient, μm/yr	SO_2 coefficient, (μm/yr)/(μg/m^3)	Number of data sets
CAMP (Haynie and Upham, 1970)	1.15 ± 0.60	0.081 ± 0.005	37
ISP (Cavender et al., 1971)	1.05 ± 0.96	0.073 ± 0.007	173
Guttman, 1968	1.79	0.024	>400
Guttman and Sereda, 1968	2.47 ± 0.86	0.027 ± 0.008	136
Chamber study (Haynie et al., 1976)	1.53 ± 0.39	0.018 ± 0.002	96
St. Louis (Mansfeld, 1980)	2.36 ± 0.13	0.022 ± 0.004	153

Note: 1 ppm SO_2 = 2620 ug/m^3.

The specified thickness of galvanized coating varies with intended use. Furthermore, the thickness of a particular coating varies considerably from one point to another. Bird's (1977) measurements revealed that 5 percent of coating thickness measurements varied from the mean by more than 46 percent. Haynie confirmed this variability with 475 thickness measurements on a single galvanized steel sheet. The life of a coating is generally proportional to its thickness; thus, rusting of the substrate steel will occur first at the thinnest spots and last at the thickest spots. As reported in Haynie (1980), The American Society for Testing and Materials (ASTM) has observed rusting at thin spots on galvanized steel wire, fencing, and sheet exposed to various types of atmospheres over many years. Some of their exposures were started in 1916 and continued until the test could reveal no additional information. In the case of sheet, the product was completely rusted and showed perforations. In general, the amount of corrosion at each site varied linearly with time. Corrosion rates at each site were calculated on the basis of time to first rust and time to complete rust for various zinc thicknesses, assuming ±40 percent thickness variability.

Originally, State College, Pennsylvania, was selected by ASTM as a control site representing a "clean" rural environment; however, the corrosion was higher there than at five other rural locations including the rural-marine environment of Santa Cruz, California, where high RH is expected to accelerate corrosion. ASTM made no pollution measurements but recognized the effects of "industrial" and "severe industrial" environments.

Zinc corroded nearly twice as fast on wire and fencing as it did on sheet, a finding that is consistent with the theoretically predicted effects of surface configuration on SO_2 deposition velocity. One would expect a greater deposition velocity onto fencing than onto sheet material. ASTM noted that fencing corrodes less near the ground than it does near the top because wind velocity increases with height, with a resultant increase in deposition velocity and similar variation of SO_2 concentration. Another factor may be the decrease in concentration near the ground, since both soil and vegetation are sinks for SO_2 (see Chapters 6 and 8).

The average corrosion rates in the Haynie (1980) study shown in Table 10-4 correspond to actual corrosion rates that are two to four times greater when the substrate is wet. Theoretical calculations indicate that the average SO_2 levels at the Pittsburgh site over the long exposure period were between 350 and 700 $\mu g/m^3$. The average at the Altoona site could have been as high as 1000 $\mu g/m^3$.

TABLE 10-4. CORROSION RATES OF ZINC ON GALVANIZED STEEL PRODUCTS EXPOSED TO VARIOUS ENVIRONMENTS PRIOR TO 1954

	Mean corrosion rate and estimated standard deviation, μm/year	
Site	Sheet	Wire and fencing
Altoona, PA	7.57 ± 0.54	--
Pittsburgh, PA	5.63 ± 0.34	10.86 ± 1.02
Sandy Hook, NJ	2.74 ± 0.30	4.37 ± 0.45
Bridgeport, CT	--	4.25 ± 0.44
Lafayette, IN	--	2.94 ± 0.34
Ithaca, NY	--	2.68 ± 0.42
State College, PA	1.27 ± 0.29	2.48 ± 0.24
Ames, IA	--	1.68 ± 0.19
College Station, TN	--	1.22 ± 0.43
Santa Cruz, CA	--	0.83 ± 0.26
Manhattan, KS	--	0.79 ± 0.27
Davis, CA	--	0.76 ± 0.42

Source: Haynie (1980).

From the relationships between theoretical and experimental studies, Haynie (1980) concluded the following:

1. Both short-term laboratory evidence and long-term exposure results for galvanized steel are consistent with theoretical considerations.
2. Damage functions for some materials can be calculated from theoretical relationships that consider factors controlling time of wetness and pollutant fluxes.
3. Wind speed and material geometry should be considered in evaluating atmospheric corrosion effects.

Harker et al. (1980) examined the variables controlling the corrosion of zinc by SO_2 and H_2SO_4 using an aerosol flow reactor. Under steady-state conditions, they made the following measurements:

Environmental measurements:

(1) Percent RH and temperature (at two points).
(2) Average flow velocity (Pitot tube).
(3) Flow velocity profile (recorded when a steady state had been established).

Aerosol measurements:

(1) Aerosol size distribution and number concentration determined at intervals during test by TSI 3050 analyzer.
(2) Two total-mass filter samples collected.
(3) Total-deposition sample collected on aluminum foil throughout each experiment.
(4) TEM deposition grid samples collected continuously.
(5) X-ray photoelectron spectroscopy samples (both zinc plate and aluminum foil) collected continuously during experiment.

Corrosion rate measurements were recorded continuously by an atmospheric corrosion monitor (ACM), which had been pretreated with either 0.1 N H_2SO_4 or ammonium sulfate. Experimental conditions were selected from the following ranges:

Temperature, °C	12-20
Relative humidity, percent	65-100
Mean flow velocity, m/sec	0.5-8
Sulfur dioxide concentration, ppb (volume)	46-216
Sulfate aerosol mass concentration, mg/m^3	1.2
Aerosol size distribution, μm diameter	0.1-1.0

In the Harker et al. (1980) study the factors controlling the rate of corrosion were RH, pollutant flux, and chemical form of the pollutant. Corrosion occurred only at RH high enough (more than 60 percent) to wet the surface; temperature did not appear to be a controlling factor within the range 12 to 20°C. The results indicate that on initial exposure SO_2-induced corrosion of zinc proceeds at a rate approximately a factor of two greater than that for the equivalent amount of deposited H_2SO_4 aerosol.

The investigators noted deposition velocities of 0.07 cm/sec for 0.1-1.0 μm SO_4^{2-} aerosols and 0.93 cm/sec for SO_2 at a friction velocity of 35 cm/sec. These factors indicate that the effects of SO_2 will dominate the effects of H_2SO_4 in most urban areas.

10.2.2.2 Paint Technology and Mechanisms of Damage--Compared with other environmental factors such as sun and precipitation, paint damage due to air pollutants is considered less important. There are at present no standard ASTM procedures for evaluating the effect of SO_2, NO_2, and/or O_3 on paints. Degradation by ultraviolet light has received the major emphasis; outdoor test stations have been located where SO_2 levels are low.

Paint erosion can be measured by loss of thickness of the paint layer, which can result from the chemical action of SO_2 and the action of light and O_3. Film erosion rates are used by paint manufacturers to determine the fail point for their formulations.

In paint formulas, the ratio of pigments to film formers is important to the overall properties of gloss, hardness, and permeability to water. If the amount of film former is too low, soiling is increased and the paint may lose the film flexibility needed for durability and become brittle. Hay and Schurr (1971) reported on the permeability of paints to water. High-permeability films are desirable for surfaces that must allow water to pass through, such as wooden exterior walls behind poorly ventilated kitchens. Low-permeability coatings are needed to protect surfaces that corrode when repeatedly moistened. The low permeability of chlorinated rubber, as well as styrene-acrylic, is advantageous for use on concrete.

Paint films permeable to water are also susceptible to penetration by SO_2 and SO_4^{2-} aerosols. The absorption of SO_2 was observed by Holbrow (1962), who found sulfites and sulfates in paint, and by Walsh et al. (1977), who used radioactive SO_2 to determine rates and saturation values for SO_2 absorption.

Concentrations of SO_2 found in fog or near industrial sites can increase the drying and hardening times of certain kinds of paints. Holbrow (1962) found that the drying time of linseed, tung, and certain castor oil paint films increased by 50 to 100 percent on exposure to 2620 to 5240 $\mu g/m^3$ (1 to 2 ppm) SO_2. The touch-dry and hard-dry times of alkyl and oleoresinous paints with titanium dioxide pigments were also reported to increase substantially; however, the exposure time of the wet films was not reported. Analysis of the dried films indicated that SO_2 had chemically reacted with the drying oils, altering the oxidation-polymerization process. No studies have been reported on the effects of SO_2 on the drying of latex paints.

Holbrow (1962) also studied the effects of SO_2 on dried paint film. In these experiments, paint films were allowed to dry, were refrigerated, and then exposed for 15 minutes to an atmosphere containing 1.2 percent SO_2. The paint films with condensed moisture were finally placed in an accelerated-weathering chamber. For all the paints except a pentaerythritol alkyd paint, the gloss decreased significantly after 1 day in the accelerated-weathering chamber. Without the accelerated weathering, the actions of SO_2 and moisture on the paint films produced only a slight reduction in gloss. Holbrow concluded that the SO_2 had sensitized the film, permitting water to be absorbed during the weathering cycle.

Bluing of green pigmented paint containing lead chromate has been observed during the early life of the film. Holbrow (1962) reproduced this effect in the laboratory by exposing the film to SO_2 and moisture and then to warmth and moisture. The bluing was probably caused by conversion of the yellow lead chromate to colorless lead sulfate. Holbrow did not attempt to correlate moisture, temperature, and pollutant concentration or to obtain dose-response data. Although very high levels of SO_2 were used, this experiment indicates that condensation and moisture evaporation are critical in concentrating the pollutant on the surface of exterior paint films; under these conditions, the film deteriorates.

Svoboda et al (1973) compared pigmented and unpigmented paint film for SO_2 permeability and found that the rate of penetration of SO_2 into a paint film was related to the pigment content. Zinc oxide and titanium dioxide pigments caused a 50- to 70-percent decrease in the rate of penetration of SO_2 into the paint film.

Spence et al. (1975) carried out a chamber study of the effects of gaseous pollutants on four classes of exterior exposure paints: oil-base house paint, vinyl-acrylic latex house paint, and vinyl and acrylic coil coatings for metals. The house paints were applied to aluminum panels by spraying. The coil coating panels were cut from commercially painted stock. The oil-base paint film was 58 μm thick; the acrylic latex, 45 μm; the vinyl coil coating, 27 μm; and the acrylic coil coating, 20 μm. The exposure chambers controlled temperature, humidity, SO_2, NO_2, and O_3. Each exposure chamber had a xenon arc lamp to provide ultraviolet radiation. A dew/light cycle was included; light exposure time was followed by a dark period during which coolant circulated through racks holding the specimens, thereby forming dew on the panels. Each dew/light cycle lasted 40 minutes and consisted of 20 minutes of darkness with formation of dew, followed by 20 minutes under the xenon arc. The total exposure time was 1000 hours. Damage was measured after 200 hours, 500 hours, and 1000 hours by loss of both weight and of film thickness. In evaluating the data, loss of weight was converted to equivalent loss of film thickness.

Visual examination of the panels coated with oil-base house paint revealed that all exposure conditions caused considerable damage. The erosion rate varied from 28.3 to 79.1 μm/year, with an average of 60 μm/year. The investigators concluded that SO_2 and RH markedly affected the rate of erosion of oil-base house paint. The presence of NO_2 increased the weight of the paint film. A multiple linear regression on SO_2 concentration and RH yielded the following relation:

$$E = 14.3 + 0.0151\ SO_2 + 0.388\ RH, \qquad (10\text{-}12)$$

where

E = erosion rate in μm/year,
SO_2 = concentration of SO_2 in $\mu g/m^3$
RH = RH in percent.

The authors reported the 95-percent tolerance limits on 99 percent of the calculated rates to be ± 44 μm/year.

Blisters formed on acrylic latex house paint at the high SO_2 levels. The blisters resulted from severe pitting and buildup of aluminum corrosion products on the substrate. The paint acted as a membrane retaining moisture under the surface and excluding oxygen that would passivate the aluminum. The vinyl coating and the acrylic coating are resistant to SO_2. The visual appearance of the vinyl coil coating showed no damage. The average erosion rate was low, 3.29 μm/year. The average erosion rate for a clean air exposure was 1.29 μm/year. The acrylic coil coating showed an average erosion rate of 0.57 μm/year.

A study of the effects of air pollutants on paint was conducted by Campbell et al. (1974). The paints studied included oil and acrylic latex house paints, a coil coating, automotive refinish, and an alkyd industrial maintenance coating. These coatings were exposed to clean air, SO_2 at 262 and 2620 $\mu g/m^3$, and O_3 at 196 and 1960 $\mu g/m^3$ (i.e., equivalent to 0.1 and 1.0 ppm of each pollutant). Other controlled study variables included light, temperature, and RH. In addition, one-half of the coatings were shaded during the laboratory exposures. Similar panels (half facing north) were exposed at field sites in Leeds, North Dakota; Valparaiso, Indiana; Research Center, Chicago, Illinois; and Los Angeles, California.

The laboratory exposure chamber operated on a 2-hour light-dew cycle (i.e., 1 hour of xenon light at 70 percent RH and a temperature of 66°C followed by 1 hour of darkness at 100 percent RH and a temperature of 49°C). Coating erosion rates were calculated after exposure periods of 400, 700, and 1000 hours. Table 10-5 summarizes the estimated erosion rates and statistical characterizations of the results. Erosion rates at O_3 or SO_2 concentrations of 0.1 ppm were not significantly different from values for clean air exposures due to high variability of the data. The erosion rates on the shaded specimens were significantly less than the unshaded panel results shown in Table 10-5; panels facing north were also less eroded. At 1 ppm pollutant concentrations, erosion rates were significantly greater than controls, with oil-base house paint experiencing the largest erosion rate increases, latex and coil coatings moderate increases, and the industrial maintenance coating and automotive refinish the smallest increases (Yocom and Grappone 1976; Yocom and Upham 1977; and Campbell et al., 1974). Coatings that contained extender pigments, particularly calcium carbonate, showed the greatest erosion rates from the SO_2 exposures. Results of field exposures also support these conclusions (Campbell et al., 1974).

TABLE 10-5. PAINT EROSION RATES AND T-TEST PROBABILITY DATA

PAINT EROSION RATES AND PROBABILITY DATA (T-TEST) FOR CONTROLLED ENVIRONMENTAL LABORATORY EXPOSURES

Type of paint	Mean erosion rate (nm/hour with 95-percent confidence limits) for unshaded panels		
	Clean air control	SO_2 (1.0 ppm)	O_3 (1.0 ppm)
House paint			
oil	5.11 ± 1.8	[b]35.8 ± 4.83	[b]11.35 ± 2.67
latex	0.89 ± 0.38	[b]2.82 ± 0.25	[a]2.16 ± 1.50
Coil coating	3.02 ± .58	[b]8.66 ± 1.19	[a]3.78 ± 0.64
Automotive refinish	0.46 ± 0.02	0.79 ± 0.66	[b]1.30 ± 0.33
Industrial maintenance	4.72 ± 1.30	5.69 ± 1.78	7.14 ± 3.56

PAINT EROSION RATES AND PROBABILITY DATA (T-TEST) FOR FIELD EXPOSURES

Type of paint	Mean erosion rate (nm/month with 95-percent confidence limits) for panels facing south			
	Rural (clean air)	Suburban	Urban (SO_2 dominant, ~60 $\mu g/m^3$)	Urban (oxidant dominant, ~40 $\mu g/m^3$)
House paint				
oil	109 ± 191	[b]376 ± 124	[a]361 ± 124	[b]533 ± 157
latex	46 ± 13	[b]76 ± 18	[a]97 ± 8	165 ± 142
Coil coating	53 ± 20	[b]254 ± 48	[b]241 ± 20	[b]223 ± 43
Automotive refinish	23 ± 28	[a]58 ± 18	41 ± 10	43 ± 10
Industrial maintenance	91 ± 41	[a]208 ± 361	168 ± 99	[b]198 ± 61

[a] Significantly different from control at an $p \leq 0.05$.

[b] Significantly different from control at an $p \leq 0.01$.

Source: Adapted from Yocom and Upham (1977).

Note: 1 ppm SO_2 = 2620 $\mu g/m^3$.

10.2.3 Fabrics

Fibers that suffer destructive action upon exposure to acids derived from SO_2 include (1) cellulosic fibers such as cotton and its close relative, viscose rayon, a regenerated cellulose, and cellulose acetate; and (2) polyamide fibers such as nylon 6 and 66. Polyester, acrylic, and polypropylene fibers are not damaged directly by SO_2; however, SO_2 concentrations can be a source of absorbed acid that can accelerate the fading of dyes and result in fabric deterioration through chemical reactions. The possibility of higher acid content due to oxidation of SO_2 to SO_3 must be considered (Salvin, 1963).

Brysson et al. (1967) exposed cotton fabrics at six different environmental sites in St. Louis, Missouri for 1 year (1963-64). The St. Louis metropolitan area sites represented industrial, urban, and suburban environments. Suspended PM in the St. Louis area was measured using periodic 24-hour hi-vol air samplers and monthly dustfall measurements. Sulfation values were determined by the lead peroxide candle method. Two fabric types were exposed in this study, a desized and scoured cotton print cloth and scoured cotton army duck. Study results indicate that there is a significant relationship between air pollution and both strength degradation and degree of fabric soiling. As shown in Figure 10-7, high pollutant levels (mean sulfation 5 mg SO_3/100 cm^2/day and/or SO_2 concentrations of 0.2 ppm or 520 $\mu g/m^3$) reduced effective fabric strength by one-sixth when compared with low pollution sites (0.5 mg SO_3/100 cm^2/day and/or 0.02 ppm or 60 $\mu g/m^3$ SO_2 concentrations). The relationship between suspended PM and fabric strength degradation was not as good as that for SO_2. No correlation between dustfall and strength degradation/effective life was demonstrated. Biological deterioration did not appear to be a major factor in this study, since a plot of fluidity values against breaking strength loss for exposed samples gave the essentially straight line function indicative of nonbiological degradation (Brysson et al. 1967).

In a review of the Brysson et al. (1967) study, Upham and Salvin (1975) report a correlation coefficient of 0.95 for breaking strength versus sulfation for cotton duck cloth. The correlation coefficient for the thinner cotton print was 0.96. Of the pollutants measured, SO_2 was most responsible for causing fabric damage (Upham and Salvin, 1975).

Zeronian (1970) carried out laboratory exposures in which cotton and rayon fabrics were exposed for 7 days to clean air with and without 250 $\mu g/m^3$ (0.1 ppm) SO_2 Both controlled environments included continuous exposure to artificial light (xenon arc) and a water spray turned on for 18 minutes every 2 hours. Loss in strength for all fabrics exposed to clean air averaged 13 percent, while the fabrics exposed to SO_2 averaged 21 percent. Zeronian et al. (1971) also exposed fabrics made from manmade fibers--nylon, polyester, and modacrylic--to controlled conditions similar to the cotton exposures, except that the SO_2 level was 486 $\mu g/m^3$ (0.2 ppm). They found that only the nylon fabrics were affected, losing 80 percent of their strength when exposed to SO_2 and only 40 percent when exposed in clean air.

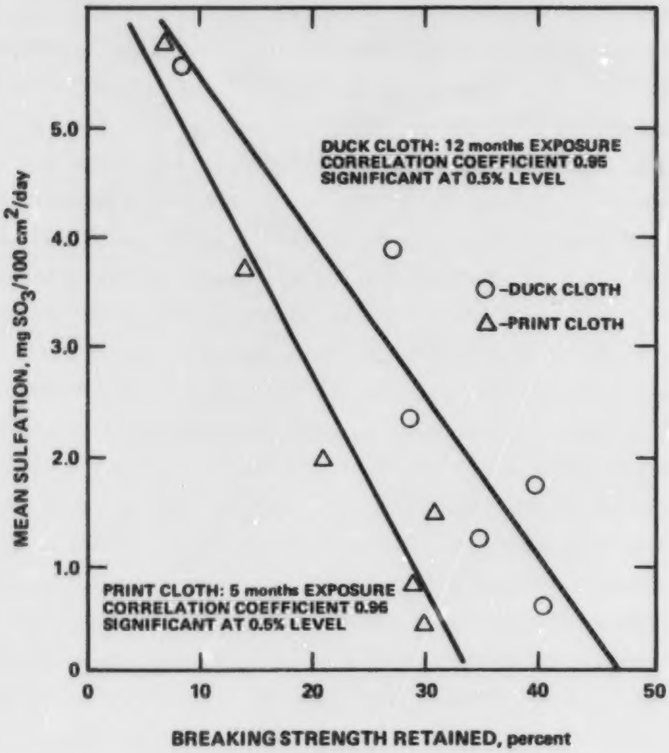


Figure 10-7. Relationship between retained breaking strength of cotton fabrics and corresponding mean sulfation rate measured at selected sites in St. Louis area.

Source: Brysson et al. (1967), Upham and Salvin (1975).

In a study designed to determine effects of air pollution on dye fading on fabrics, 67 dye-fabric combinations were tested. The test samples were exposed in the dark at 11 sites representing climatic regimes and urban/rural conditions. High temperatures and humidities appeared to increase fading in the presence of air pollutants. Urban sites produced significantly higher fading than rural sites. Fading was highest at the site with the highest SO_2 and TSP concentrations (Beloin, 1972); however, a more rigorous test of the significance of SO_2 to dye fading suggests that the only correlation between SO_2 and dye fading found in the Beloin study may not have represented a cause-and-effect situation. Results of a two-level factorial chamber experiment exposing various dyed fabrics to SO_2 at .03 ppm or .50 ppm (79 and 1310 $\mu g/m^3$, respectively) and two levels of NO_2 and O_3 showed no significant fading attributable to SO_2 (Upham et al., 1975).

10.2.4 Building Materials

The deterioration of inorganic building materials occurs initially through surface weathering. Moisture and salts are considered the most important factors in building material damage. Many researchers believe that the mechanism of damage from air pollution involves the formation of salts from reactions in the material; subsequently, these surface salts dissolve in moist air and are washed away by rainfall. The components of inorganic building materials can react with SO_2 and sulfates (Luckat, 1972; Winkler, 1975; Arnold et al. 1976). Luckat (1977) reported good correlation of damage to stone with rate of SO_2 uptake. Other researchers believe that the role of air pollutants in stone and concrete damage has frequently been overestimated (Riederer, 1974; Niesel, 1979). These authors report that inorganic building material damage is dominantly associated with RH >65 percent and freeze/thaw weathering. Some researchers indicate that microorganisms must also be considered in order to quantify damage to building materials due to ambient pollutant concentrations (Winkler, 1966; Riederer, 1974; Krumbein and Lange, 1978; Eckhardt, 1978; Hansen, 1980). Sulfur chemoautotrophs are well known for the damage they can cause to inorganic materials. These microorganisms (e.g., Thiobacillus) convert reduced forms of sulfur to H_2SO_4 (Anderson, 1978). Presence of sulfur-oxidizing bacteria on exposed monuments has been confirmed (Vero and Sila, 1976). The relative importance of biological, chemical, and physical mechanisms, however, have not been systematically investigated. Thus, damage functions definitely quantifying the relationship of pollutant concentrations to stone and concrete deterioration are not available in the literature. Air pollution damage to glass is also not quantifiable at present (Newton, 1974).

10.2.4.1 Stone--Niesel (1979) completed a literature review on the weathering of building stone in atmospheres containing SO_x, which includes references from 1700 to 1979. In brief, he reports that weathering of porous building stone containing lime is generally characterized by accumulation of calcium sulfate dihydrate in the near-surface region. The effect of atmospheric pollutants on the rate of weathering is believed to be dominantly controlled by the stone's permeability and moisture content. Migrating moisture serves primarily as a transport medium. Sulfur dioxide is sorbed and thus can be translocated internally while being oxidized to sulfates. Reacting components of the building stone are thus leached, the more soluble compounds inward and the less soluble toward the surface, often forming a surface crust.

Sengupta and de Gast (1972) also report that SO_2 sorption causes physical changes in stone involving changes in porosity and water retention. Removal of calcium carbonate changes the physical nature of the stone surface. The hard, nonporous layer that forms as a result of alternate freezing and thawing may blister, exfoliate, and separate from the surface. If the stone contains some substances that are unaffected by SO_2, the surface can deteriorate unevenly. The conversion of calcium carbonate to calcium sulfate results in a type of efflorescence called "crystallization spalling."

According to Gauri (1979) acidic precipitation also contributes to the weathering process of building stone. Gauri reported that marble that is directly exposed to rainfall undergoes nearly continuous erosion as the acid dissolves the calcium carbonate, allowing calcite granules to break away and wash off.

At a recent NAS conference on conservation of historic stone monuments and buildings, Tombach (1981) summarized mechanisms contributing to stone decay (see Table 10-6). While more information on these mechanisms is becoming available, the relative contribution of ambient SO_2 to deterioration of the various types of stone is not yet clear.

10.2.4.2 Cement and Concrete--Portland cement, the major active constituent of most concrete, is manufactured by the high-temperature reaction of a mixture of limestone, alumina, silicates, and iron salts found in clay. Cement, the binding agent in concrete, is an alkaline material that reacts with SO_2 and thus also suffers erosion and spalling effects.

The chemical action of SO_2 or sulfates on cement or concrete can be described as a dual mechanism. Calcium hydroxide in cement and concrete can be converted to calcium sulfate, which reacts to form calcium sulfate aluminum hydrate (ettringite), with a substantial increase in volume. Cement for dams and culverts requires special formulation for SO_4^{2-} resistance when exposed to SO_4^{2-} concentrations > 200 ppm in water (Nriagu, 1978).

Litvin (1968) examined concrete samples containing Portland cement and marble aggregate with sand at an industrial site in Buffington, Indiana. Some changes were noted in the marble aggregate, but a more observable change was found in the cement portion. Sealants were evaluated as protective coatings; their use was accompanied in some cases by surface efflorescence.

Recently, specialized concretes in which sulfur replaces cement as the binding agent have been successfully tested for resistance to acid and salt corrosive environments. These sulfur

TABLE 10-6. MECHANISMS CONTRIBUTING TO STONE DECAY. PRINCIPAL ATMOSPHERIC FACTORS PARTICIPATING IN THESE MECHANISMS ARE DENOTED BY SOLID CIRCLES; SECONDARY FACTORS ARE INDICATED BY SOLID TRIANGLES

Mechanism	Rainfall	Fog	Humidity	Temperature	Insolation	Wind	Gaseous Pollutants	Aerosol
A. External Abrasion								
1. Erosion by windborne particles						●		●
2. Erosion by rainfall	●							
3. Erosion by surface ice	●	●		●				
B. Volume Change of Stone								
1. Differential expansion of mineral grains				●			▲	
2. Differential bulk expansion due to uneven heating				●	●			
3. Differential expansion of differing materials at joints				●				
C. Volume Change of Material in Capillaries and Interstices								
1. Freezing of water	●	●		●				
2. Expansion of water when heated by sun	●	●		●	●			
3. Trapping of water under pressure when surface freezes	●	●		●				
4. Swelling of water-imbibing minerals by osmotic pressure	●	●	●				▲	▲
5. Hydration of efflorescences, internal impurities, and stone constituents	●		●				▲	▲
6. Crystallization of salts			●	●	●	●	▲	▲
7. Oxidation of materials into more-voluminous forms	●	●					▲	
D. Dissolution of Stone or Change of Chemical Form								
1. Dissolution in rain water	●			●			●	●
2. Dissolution by acids formed on stone by atmospheric gases or particles and water	●	●	●	●			●	●
3. Reaction of stone with SO_2 to form water-soluble material	●	●		●			●	
4. Reaction of stone with acidic clay aerosol particles	●	●		●				●
E. Biological Activity								
1. Chemical attack by chelating, nitrifying, sulfur-reducing, or sulfur-oxidizing bacteria			●	●			●	
2. Erosion by symbiotic assemblages and higher plants which penetrate stone or produce damaging excretions	▲	▲	●	●				

Source: Tombach (1981).

concretes, developed by the Federal Bureau of Mines, displayed good resistance to both acid and salt attack and to damage by freeze-thaw cycling. (Sulphur Institute, 1979; McBee and Sullivan, 1979; and Sullivan and McBee, 1976).

10.2.5 Electrical Equipment and Components

Robbins (1970) and ITT Electro-Physics Labs (1971) studied the damaging effects of SO_2 and particles on electronic components and estimated the cost of this damage. The ITT Electro-Physics Labs report considered damage to 11 categories of electronic components for which a literature survey indicated that SO_2 pollution would be mainly responsible. Information gained directly from manufacturers, however, indicated that particles were the major factor in degradation and failure of electronic components and equipment (see Section 10.3.1). Reduction of SO_2 and particulate concentrations would have little effect on costs for the prevention of corrosion; the same protective measure would still be required for low concentrations of pollutants. Since the cost of corrosion-resistant metals is far outweighed by the expense of equipment failure, they are used even in environments where air pollution is minimal.

10.2.6 Paper

Modern papers are manufactured from cellulose. On exposure to acids, paper is hydrolyzed and loses strength. Analyses of paper from old books have shown H_2SO_4 content on the edges up to nearly 1.5 percent, with differences between the amounts in the edges and the center of a page sometimes approaching 1 percent (Parker, 1955). Spedding et al. (1971), studying the mechanism in work with radioactive labeling techniques, determined that SO_2 is readily absorbed by paper and oxidized to H_2SO_4 by the metallic impurities in the paper. The reaction may also involve the lignins in the paper, resulting in the formation of lignosulfonic acids. Walsh et al. (1977) showed that SO_2 is rapidly absorbed by uncoated wallpaper and less rapidly absorbed by vinyl-coated paper. Although most paper is used in objects with a short service-life, and steps have been taken to improve the acid resistance of books, the preservation of documents is of concern in museums and archives. Coating paper with polymers impervious to gases is an established process. It is estimated that 50 percent of the books printed between 1900 and 1940 are in need of conservation. The New York Public Library conserves old books by microfilming, lamination, and electrostatic reproduction (Kingery, 1960). The library spent $900,000 between 1952 and 1967 to microfilm books that had deteriorated (Waddell, 1974).

10.2.7 Leather

Leather has a high capacity for absorbing SO_2. Spedding et al. (1971) reported that the controlling factor in SO_2 uptake is the rate of SO_2 diffusion to the leather surface. The formation of H_2SO_4 in the presence of water is followed by hydrolysis of the protein (collagen) of which leather is principally composed. This weakening of leather causes cracking and ultimately results in reduction of the leather to a red-brown powder (Spedding et al., 1971; Yocom and Grappone, 1976).

The destruction of leather by absorption of SO_2 has long been known and was described in detail by Prenderleith (1957). The buildup of H_2SO_4 in aged leathers correlates with deterioration, which can be reduced by inactivating the SO_4^{2-} ion and by pH buffering. Deterioration of leather is important in bookbinding and in leather upholstery. The use of artificial leathers has reduced damage costs.

10.2.8 Elastomers and Plastics

The deterioration of natural rubber and synthetic elastomers under weathering conditions has been studied extensively. Heat, light, oxygen, certain metallic ions, and particularly O_3 cause deterioration; but the literature does not mention SO_2 damage to rubber. Rubber, in fact, is used as an acid-resistant coating. The problem of determining ambient air pollution effects on rubber is complicated by the presence of O_3, which attacks the double bonds in both natural rubber and the butadiene-styrene and butadiene-acrylonitrile synthetics.

Haynie et al. (1976) conducted a chamber study on rubber to determine the effects of O_3, SO_2, and NO_2 under controlled conditions of temperature, humidity, and light. Exposures were made at concentrations of 0.1 and 1.0 ppm for each pollutant (in $\mu g/m^3$; 262 and 2620 for SO_2, 196 and 1960 for O_3 and 188 and 1880 for NO_2). As expected, O_3 was responsible for accelerated cracking of the rubber. Sulfur dioxide did not have any effect.

Verdu (1974) presented a theoretical study of the effect of air pollutants on the weathering of plastics. He attributed a direct deteriorating effect on plastics to O_3 and suggested that air pollutants such as SO_2 may form active compounds through photochemical reactions leading to oxidation chain reactions. In light-exposure trials, SO_2 increased the rate of degradation of polystyrene.

10.2.9 Works of Art

Although works of art are composed of materials already discussed in earlier sections of this chapter, they are briefly treated here as a separate category because the cost of the materials involved does not represent the cost of the item.

The deteriorating effects of SO_2 and particles are well known to museum conservators whose function is to preserve and restore works of art. The rate of pollutant-related deterioration has increased markedly in the last 50 years. The damage is striking in Europe, where ancient buildings, paintings, frescoes, stained glass windows, bronze sculptures, and marble statuary have suffered deterioration.

Newton (1974) has investigated the cause of deterioration of medieval stained glass windows. He found that the main cause of decay is the leaching of potassium ions from the silicate glass by condensed water. Another cause is SO_2, which produces opaque white crusts containing $CaSO_4 \cdot 2H_2O$ and syngenite ($K_2Ca(SO_4)_2 . H_2O$). The poor durability of medieval glass is due to its high content of alkaline earths such as lime and magnesia.

Riederer (1974) conducted a study of the corrosion of bronze sculpture by air pollutants. Sulfates were found in the corroded surfaces.

The sandstone of the cathedral in Cologne, located in a highly polluted urban area, has suffered serious erosion due to the reaction of sulfur acids with calcium carbonate that form calcium sulfate, which is leached out by rain (Luckat, 1972). Decay and deterioration of the Taj Mahal in India has also been attributed to SO_2 and other pollutants emitted from a nearby petrochemical complex (Gajendragakar, 1977).

Damage occurring in Venice, Florence, Rome, Athens, London, and Cologne has been attributed to the effect of SO_2 from industrial areas in these cities (Yocum and Upham, 1977). The United States is also concerned about the deterioration of public buildings and monuments. The National Bureau of Standards (NBS) was asked by the National Park Service to investigate methods for preservation of stone after erosion was noted in the facade of the Lincoln Memorial in Washington, D.C. (Sleater, 1977).

Sleater (1977) investigated damage to stone from the action of SO_2, salt, sodium sulfate, and light. Conservation materials including epoxy resins, fluorosilicates, and silicone resins were evaluated. The conservation methods recommended to the National Park Service varied with the exposure conditions.

The damage to the Acropolis caused by SO_2 and SO_3 has resulted in a massive interdisciplinary effort by the Greek government to protect the ancient buildings from further deterioration (Yocom, 1979).

10.2.10 Review of Damage Functions Relating Sulfur Dioxide to Material Damage

Even the most reliable damage functions must be used with caution. As noted by Sereda (1974), more data are required to take account of orientation, location, and design of materials in use. Those listed in Table 10-7 were selected on the basis of their treatment of independent variables and their inclusion in major literature reviews. As discussed previously (see Section 10.2.1.2.2.2) damage functions vary in form, reflecting different parameters measured and methods of measurement. Time-of-wetness (often expressed as RH above a critical value) is the most important variable in these damage functions.

Functions for zinc or galvanized steel appear to show the best fit, followed by the functions for oil-based house paint. The field studies by Haynie and Upham (1970) and Haynie (1980) and chamber study by Haynie et al. (1976) incorporated critical variables and provided relatively reliable damage functions for galvanized steel. The functions selected for weathering and enameling steel and for oil-based paint also utilized these critical environmental variables.

When these functions are used to estimate damage, other factors must be considered, such as the amount of exposed (uncoated) metal, the percentage of buildings with oil-based (not Latex) paint, and temperature and humidity variables (sites in the arid southwest compared with sites in the relatively humid northeast).

TABLE 10-7. SELECTED PHYSICAL DAMAGE FUNCTIONS RELATED TO SO_2 EXPOSURE

Material	Reference	Exposure-Response relationships	R^2
Zinc	Haynie and Upham, 1970	$Y = 0.001028\ (RH - 48.8)\ SO_2$	0.92
Galvanized steel	Haynie et al., 1976	$corr = (0.0187\ SO_2 + e^{41.85 - 23,240/RT})t_w$	0.91
Galvanized steel	Haynie, 1980	$corr = 2.32\ t_w + 0.0134v^{0.781}SO_2t_w$	Not provided by author.
Oil-base house paint	Spence et al., 1975	$Y = 14.3 + 0.0151\ SO_2 + 0.388\ RH$	0.61
Enameling steel	Haynie and Upham, 1974	$corr = 325\ \sqrt{t}\ e^{[0.00275\ SO_2 - (163.2/RH)]}$	Not provided by authors.
Weathering steel	Haynie et al., 1976	$corr = [5.64\ \sqrt{SO_2} + e^{(55.44 - 31,150/RT)}]\ \sqrt{f_w}$	0.91

corr = depth of corrosion or erosion, μm

Y = corrosion/erosion rate, μm/yr

SO_2 = $\mu g/m^3$ SO_2

R = gas constant (1.9872 cal/gm mol K)

RH = percent annual average relative humidity

f_w = fractional time of panel wetness

t_w = time of wetness in years

v = wind velocity in m/s

T = geometric mean temperature of panels when wet, K

t = time exposure, years

Note: 1 ppm SO_2 = 2620 $\mu g/m^3$.

10.3 PARTICULATE MATTER

A report by the National Research Council (1979) on airborne particles noted that deposition of dust and soot on building materials not only significantly reduces their aesthetic appeal, but also, either alone or in concert with other environmental factors, results in direct chemical attack. Because of the paucity of data (see Chapter 5) regarding TSP size distribution and composition, it is difficult to determine the specific types of particles and chemical constituents that have damaged or soiled a particular structure. As discussed below, chemical composition of PM is highly important to its corrosiveness. Particle size may be important; research is under way on the role of particle size in soiling of paint.

10.3.1 Corrosion and Erosion

Early studies indicated that suspended PM plays a significant role in metal corrosion. Sanyal and Singhania (1956) wrote that the influence of suspended PM was "profound." They ascribed the corrosive effects of particles to (1) electrolytic, hygroscopic, and/or acidic properties and (2) their ability to sorb corrosive gases (e.g., SO_2). Chandler and Kilcullen (1968) pointed out that it is difficult to predict corrosion rates separately for SO_2 and PM since they frequently coexist at high levels. Other field studies have not established a conclusive correlation between total suspended PM and corrosion though analysis of data continues (Mansfeld, 1980; Haynie and Upham, 1974; and Upham, 1967).

Moist air polluted with SO_2 and PM results in a more rapid corrosion rate than air polluted with SO_2 alone (Yocom and Grappone, 1976; Johnson et al., 1977). Kottori (1980) observed that zinc and galvanized steel corrosion rates appeared to be related to the SO_4^{2-} content of TSP. Chloride content of dust also may contribute to accelerated corrosion of steels (Gibbons, 1970; Bresle, 1976).

Barton (1958) reported that dustfalls contribute to the initial stages of metal corrosion but that their influence becomes less important as a layer of rust forms. Two classes of PM: hygroscopic salts (including those of natural origin such as sodium chloride) and acid smut appear to be definitely associated with corrosion.

A review of atmospheric factors affecting metal corrosion provides evidence of a relationship between salinity and corrosion (Guttman and Sereda, 1968). Corrosion of metals can be accelerated by deposition of particles that are hygroscopic and therefore increase surface wetness time. The influence of hygroscopic substances on metal corrosion rates has been previously discussed in Section 10.2.1.1.4. As also noted in Section 10.2, particles can disrupt the protective oxide films formed on metal surfaces such as nickel, copper, aluminum and stainless steel, resulting in pitting (Russell, 1976; NRC, 1979). Russell (1976) noted that airborne particles often play an important role in the attack by SO_x on electrical contact surfaces by acting as points for the concentration of active ionic species. Smoothly polished or electroplated surfaces are less likely to retain solid deposits originating from airborne particles (Larrabee, 1959).

Acid smut is a highly corrosive, sticky material formed in and emitted mainly from furnaces, notably in power plants that burn liquid fuels containing sulfur (Ireland, 1968). This material would not usually be considered suspended PM, as it occurs as agglomerates of carbon, ash, and H_2SO_4 up to 0.5 cm or more in diameter that fall close to the source (Rowden, 1968; Potter, 1971). According to Hoshizawa and Koyata (1970), acid smut mainly settles out within 400 meters of the source under conditions of light wind.

Japanese investigators analyzed a large (>10 drums [sic]) sample of acid smut and found the H_2SO_4 content to be 30 percent (Oyama et al., 1974). Damage to painted surfaces, automotive finishes, and even agricultural crops can be substantial. As noted in a review of residual oil firing problems, "public reaction can be quite severe" (Exley, 1970). A report on the status of "public nuisances" in the electric power industry of Japan reported progress in determining the cause of acid smut and in developing preventive techniques (Overseas Public Nuisance Study Mission, 1965).

Finishes on automobiles parked near industrial sites have often been severely damaged. Staining and even pitting of auto finishes have been traced to iron particles from nearby plants. Cars parked near brick buildings being demolished have been damaged by alkali mortar dust during humid weather. Damaged auto finishes had to be repainted because color changes were not reversible by washing or polishing (Fochtman and Langer, 1957).

Parker (1955) reported that numerous black specks accumulated on freshly painted buildings in industrial areas. The building exteriors became distinctly soiled and required cleaning or repainting in 2 or 3 years, depending on PM concentrations in the air. When PM became embedded in the paint film, the coating was both esthetically and physically damaged. Embedding of particles provides nucleation sites at which other pollutants can concentrate. Cowling and Roberts (1954) suggest that particles promote the chemical deterioration of paint by acting as wicks to transfer the corrosive SO_2 solution to the underlying surface. Luckat (1972) suggested that dusts containing heavy metals may accelerate stone erosion by catalytic effects on the oxidation of ambient SO_2 to H_2SO_4.

10.3.2 Soiling and Discoloration

Soiling is the accumulation of PM on the surface of a material. As defined by Faith (1976), soiling arises from the deposition of particles less than 10 μm by impingement on surfaces which then mingle with settled dust. Soiling produces a change in reflectance from opaque materials and reduces light transmission through transparent materials (Beloin and Haynie, 1975; NRC, 1979). Soiling due to airborne particles from manmade sources results in increased cleaning costs for building and other materials and in reduction in the useful life of fabrics.

Considerable uncertainty exists about the level of accumulated PM that leads to increased cleaning, since not only are socioeconomic factors involved in a decision to clean, but the accumulation of particles must first be perceived as soiling. Carey (1959), discussing household cleaning, observed that, although in a room with dusty air, particles descended continuously onto paper, the paper remained apparently clean for a period of time and then, almost suddenly, appeared dirty. The dingy appearance occurred when the paper surface was covered with dust specks spaced about 10 to 20 diameters apart. The question of least perceptible cover concerns the sensitivity of the eye and can be evaluated by use of patterns as shown in Figure 10-8. If the patterns in this figure are viewed at a distance too great to perceive the individual dots it will be found that when the contrast is strong, e.g., black on white, it is possible to distinguish a clean circle from its surroundings when 0.2 percent of the area is covered with dots; with a weaker color contrast 0.4 percent is the limiting coverage. The effect is subjective, and it is not easy to judge between coverages that differ by a factor of $\sqrt{2}$. The deposition of relatively large particles on flat, horizontal surfaces noted in Carey's report applies more to household soiling than to the soiling of outside, vertical structures.

Following the concept that dinginess or dustiness is a visually perceived phenomenon, Hancock et al. (1976) conducted a quantitative experiment on dustiness. In support of Carey's observations, these authors found that, with maximum contrast, a 0.2 percent effective area coverage (EAC) by dust can be perceived against a clean background. The minimum perceivable difference between varying gradations of shading was a change of about 0.45 percent EAC. The results also revealed that a dust deposition level of 0.7 percent EAC was required before the object covered in dust was deemed unfit to use. A telephone survey indicated that the minimum tolerable interval between household dustings was every 4 days. Combination of the telephone survey information with the level of objectionable dust coverage implied that a dustfall rate of less than 0.17 percent EAC/day would be tolerable for the population at large. Based on these evaluations and assuming an atmospheric particle density of 2 g/cm^3, an average projected area radius of 5 µm for the settled particles, the total mass flux which corresponds to 0.17 percent EAC per day was estimated to be 1.36 mg/cm^2 mo. (roughly 40 tons/mi^2 mo).

Esmen (1973), found that exposed surfaces reached an EAC of 0.4 percent seven times faster in an urban location than in a rural location. Ambient particle concentrations, however, were not reported.

10.3.2.1 Building Materials--Under high wind conditions, large particles entrained in the windstream tends to cause a slow erosion of surfaces similiar to sandblasting. Particles also fill surface pores of many sandstones causing them to become uniformly darkened.

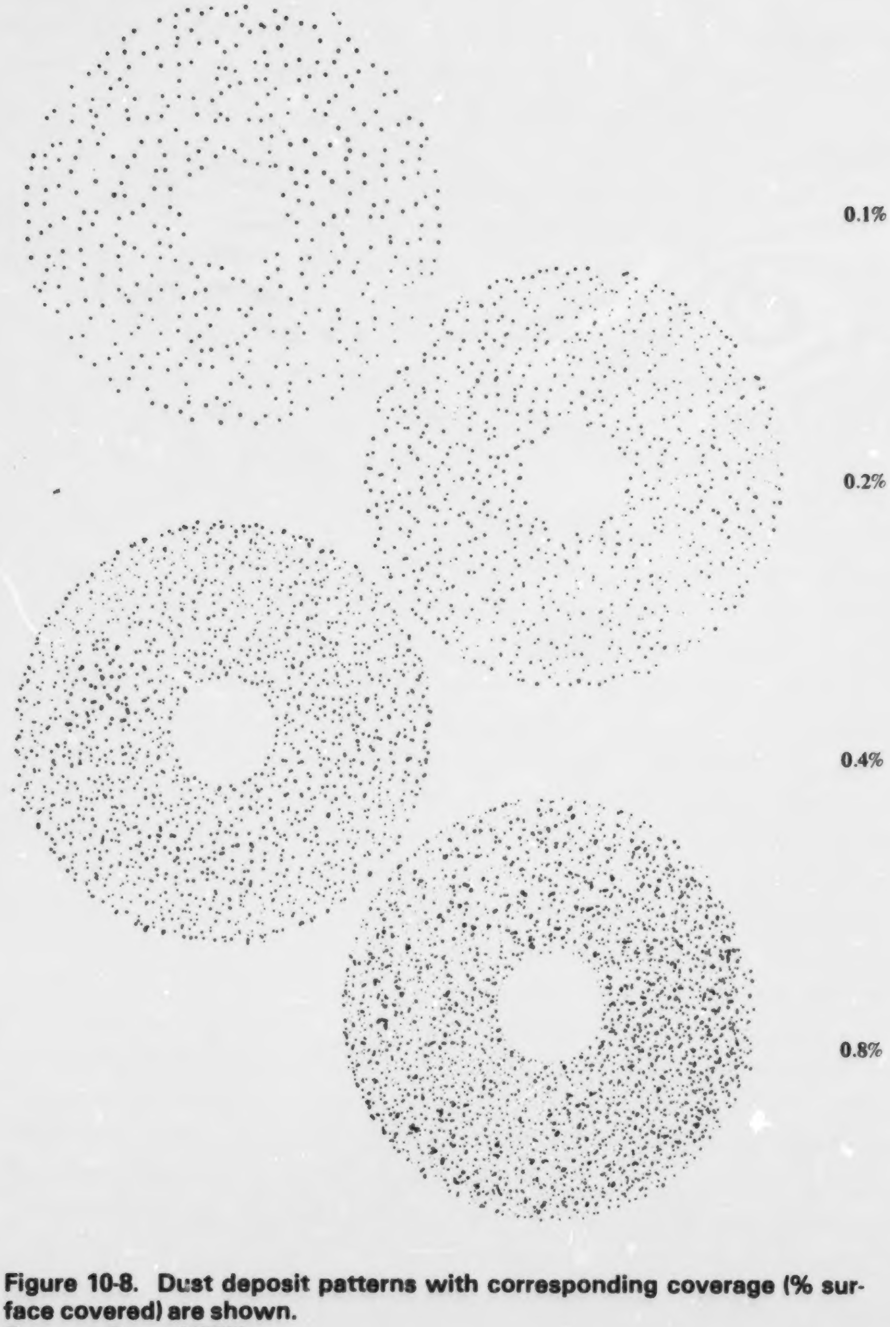


Figure 10-8. Dust deposit patterns with corresponding coverage (% surface covered) are shown.
Source: Carey (1959).

Particles can contribute to chemical decay of marble, limestone, and dolomite stone work (see Table 10-6, Section 10.2.4), and concrete structures if they carry acids and soluble salts (NRC, 1979). Scanning electron microscopy (SEM) examination of exposed marble from several locations in northern Italy provides evidence of a major role for carbonaceous particles in marble deterioration. Del Monte et al. (1981) report that SEM morphological characterization, together with particle analysis by X-ray diffraction and X-ray energy dispersive analyzer data, identifies the majority of the carbonaceous particles as products of oil fired boiler/combustion. Particle median diameter was ~ 10 μm. Del Monte et al. also presented evidence strongly suggesting that the carbonaceous particles are very important in oxidizing SO_2 to SO_4^{2-}. If further research substantiates this finding, current understanding of the relative roles of SO_2 and PM in marble deterioration might well be substantially modified.

Beloin and Haynie (1975) developed dose-response relationships for suspended PM and soiling of various building materials. They compared the rates of soiling by different levels of TSP on six different building materials over 2 years. The mean annual PM concentrations at the five study sites ranged from 60 $\mu g/m^3$ for a rural residential location to 250 $\mu g/m^3$ for an industrial residential environment. The exposed materials included painted cedar siding, concrete block, brick, limestone, asphalt shingles, and window glass. Table 10-8 shows the results of regression for soiling of building materials as a function of TSP exposure.

10.3.2.2 Fabrics--Although PM obviously soils fabrics, researchers have noted that it is only damaging when the particles are highly abrasive and the fabrics are frequently flexed. Curtains hanging in open windows, serving as filters in polluted areas, provide a good example. Weakened as a result of such exposure, curtains often split in parallel lines along the folds (Yocom and McCaldin, 1968.) The more tightly woven the cloth, the more resistant it is to soiling (NRC, 1979).

Because of soiling, fabrics may be washed more often. Excessive washing may reduce fabric strength, leading to a poorer appearance and to shortened life. Sunlight, water vapor, SO_x, NO_x and O_3 concentrations, however, are believed to have a more significant effect on the service life of fabrics. Insolation decoloration is considered to be the most important service life reduction factor (NRC, 1979).

10.3.2.3 Household and Industrial Paints--Exterior paints can be soiled by liquids and solid particles composed of soot, tarry acids, and various other constituents. Beloin and Haynie (1975) determined by reflectance measurements that the degree of soiling of painted surfaces was directly proportional to the square root of the PM dose, accounting for 74 to 90 percent of the measured variability. As one example, the linear regression results of soiling of acrylic emulsion house paint by exposure to suspended PM is shown below:

(720 samples, $R^2 = 0.902$)

$$Y = B\sqrt{(\text{TSP x months of exposure})} + A \qquad (10\text{-}13)$$

TABLE 10-8. RESULTS OF REGRESSION FOR SOILING OF BUILDING MATERIALS AS A FUNCTION OF TSP EXPOSURE

Material	N	A	B	S_A^2	S_B^2	S_E^2	R^2
Oil base paint	400	89.43	-0.2768	0.0641	0.000069	7.6510	0.745
Tint base paint	400	86.13	-0.2618	0.0571	0.000061	6.8265	0.738
Sheltered acrylic emulsion paint	400	91.54	-0.593	0.1156	0.000123	13.8143	0.880
Acrylic emulsion paint	720	90.79	-0.4131	0.0497	0.000026	8.3791	0.902
Shingles	48	43.50	-0.199	0.5771	0.000258	7.6992	0.769
Concrete	160	41.75	-0.0458	0.1338	0.000080	7.5011	0.143
Coated limestone	80	44.57	+0.0779	0.2464	0.000164	6.9046	0.347
Uncoated limestone	80	46.99	-0.0503	0.1500	0.000089	4.2035	0.266
Coated red brick	80	12.95	-0.0296	0.0223	0.000013	0.6255	0.459
Uncoated red brick	80	14.88	-0.0374	0.0331	0.000020	0.9274	0.477
Coated yellow brick	80	45.05	-0.1133	0.5337	0.000317	14.9533	0.342
Uncoated yellow brick	80	43.21	-0.1133	0.2740	0.000168	7.6773	0.503
Glass	45	0.2806	+0.0314	0.008077	0.000007	0.6851	0.340

Note: Equation used in this regression analysis was reflectance = $B\sqrt{(\text{TSP} \times \text{months of exposure})} + A$,

except in the case of shingles, where reflectance = $\frac{B(\text{TSP} \times \text{months of exposure})}{10} + A$

N, = Number of data sets (dependent upon the number of controlled variables in the factorial experiment).

A, = Intercept of linear regression.

B, = Slope of linear regression.

S_A^2, = Estimated variance of intercept.

S_B^2, = Estimated variance of slope.

S_E^2, = Residual variance (error).

R^2, = Correlation index (fraction of variability accounted for by regression).
TSP = Total suspended particulate matter in $\mu g/m^3$.

Source: Abstracted from Beloin and Haynie (1975).

Where: Y = Measured percent reflectance (Photvolt Model 625)

B = Slope of linear reflectance

TSP = Average TSP concentration ($\mu g/m^3$)

A = Intercept of linear regression

t = Exposure time (months)

Based on this equation, Figure 10-9 summarizes the soiling of acrylic emulsion house paint as a function of exposure time and particle concentrations. Although it is recognized that socioeconomic factors control ability and motivation to maintain clean surfaces (Beloin and Haynie 1975), repainting frequencies can be estimated. Assuming an individual responds to a defined change in the reflectance of house paint by repainting, that person will repaint a house twice as often in an environment with a TSP concentration of 260 $\mu g/m^3$ in comparison to 75 $\mu g/m^3$. Specifically, at a 35-percent change, this houseowner would repaint every 4 years when TSP is 75 $\mu g/m^3$ and every 2 years when TSP is 260 $\mu g/m^3$.

10.4 SUMMARY, PHYSICAL EFFECTS OF SULFUR OXIDES AND PARTICULATE MATTER ON MATERIALS

Sections 10.2 and 10.3, indicate that the best data base for evaluating the association between SO_x and materials damage is that which concerns corrosion of metal. The parallel case for PM is soiling. Of the damage functions developed for corrosion of metals by SO_x, the one for zinc appears to show the best fit. Relationships between SO_x exposure and corrosion of other metals are slightly less well established. There is evidence that PM aggravates corrosion, especially when coexisting with SO_x. This is most likely due to the hygroscopic salt content of the PM. No general mathematical expression of the relative contribution of PM to corrosion of metal, however, has been established.

Both PM and SO_x damage paint. Damage functions for erosion of various paints by SO_x and soiling of paint by PM have been developed. The varying properties of the several types of paint used on exposed materials make it difficult to construct mathematical expressions for effect of PM, SO_2, or both on all paints.

Building materials also are eroded by SO_x and soiled by PM. That damage is attributable to both pollutants is well established; quantitative general relationships between ambient concentration and effect are lacking, however. For erosion of building materials, particularly stone and concrete, the contribution of sulfur oxides relative to other agents is not clear. As for soiling, although some damage functions have been developed, the lack of understanding of the role of particle size and composition makes it difficult to generalize about soiling effects of all suspended PM on all building materials.

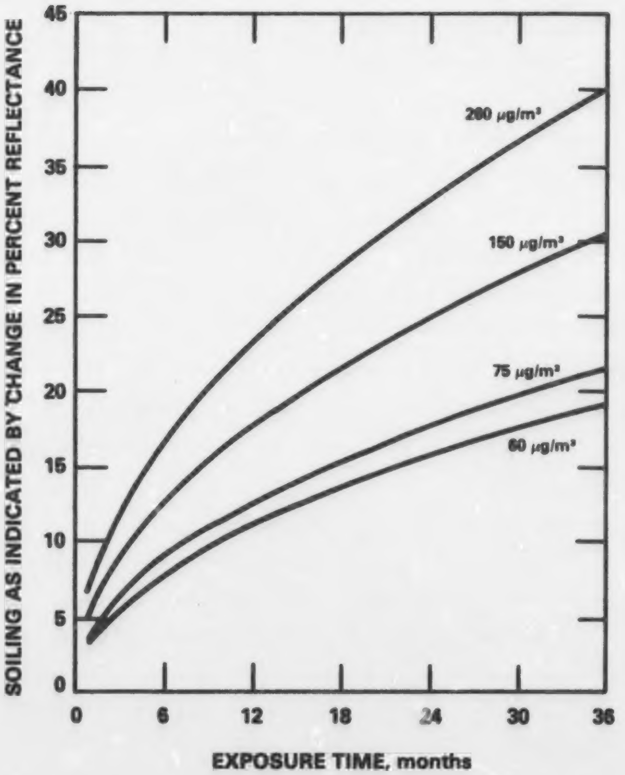


Figure 10-9. Representation of soiling of acrylic emulsion house paint as a function of exposure time and particle concentrations.

Source: Abstracted from Beloin and Haynie (1975).

Data on the effects of PM, SO_x, or both on other surfaces are even less well established. Some evidence shows damage to fabrics, leather, paper, glass, plastic, and works of art composed of one or many materials, but this evidence is largely qualitative and sketchy. It is difficult to develop reliable estimates of effects attributable to specific ambient pollutant concentrations for these materials.

10.5 ECONOMIC ESTIMATES

10.5.1 Introduction

Several types of financial losses result from damage and soiling:

1. Reduced service life of a material.
2. Decreased utility of a material.
3. Use of suitable substitute materials.
4. Losses due to an inferior substitute.
5. Protection of susceptible materials.
6. Additional required maintenance, including cleaning.

The major losses of amenity, as defined by Mäler and Wyzga (1976), are associated with enduring and suffering soiled, damaged, or inferior products and materials because of pollution, in this case, PM, SO_x, or both. In addition, amenity losses are suffered when pollution damage repair or maintenance procedures result in inconvenience or other delays in normal operations. Some of these losses, such as effects on monuments and works of art, are especially difficult to specify (Mäler and Wyzga, 1976).

The reduced value and attractiveness of property and the extra costs of cleaning and maintenance resulting from air pollution levels must obviously be considered when assessing financial losses. In addition to the diseconomy of property value losses, the consumer is also burdened with less directly measurable psychological distresses encountered when experiencing pollution nuisances. Less tangible externalities such as these have an impact on consumers as real as any directly measurable financial losses.

In calculating monetary damage, the approach selected depends on whether financial losses or losses of amenity are emphasized, the type of damage being considered, and the availability of cost information. The literature on pollutant effects on materials has been dominated by calculations of financial loss based on the physical damage function approach. This approach leads to error in financial loss estimates for the reasons listed below:

1. Reliable information on physical damage is not available for all economically important materials.
2. Information is lacking on the spatial and temporal distribution of the materials being used.
3. Techniques do not reflect the use of resistant materials that last longer and require less maintenance.

4. Estimates assume that galvanized steels are left uncoated.
5. Many materials may wear out before they are significantly damaged by pollutants.
6. Many loss estimates assume that substitute materials cost more than the original material, and that the cost differential is attributable to pollution.

Due to increased costs of labor, many materials are being sought that reduce maintenance and also resist pollutant attack. Savings realized primarily from lower maintenance should be deducted from costs charged to pollution (Glass, 1978).

The estimation of monetary damage associated with soiling is not dominated by the physical damage approach. In these studies, loss of amenity has been considered as well as direct financial loss, since socioeconomic variables are heavily involved. The approaches reflect the shift in emphasis: Nonmarket and indirect market approaches have been used to relate changes in air quality to changes in the amount of money people are willing to pay for improved air quality (reduced air pollution). This relationship can be documented through a survey of affected individuals or development of relationships between environmental quality and available data on price differentials. A major source of error is that these approaches demand that all factors that affect cost other than air quality must be accounted for. It is also difficult to distinguish among the effects of many different air pollutants.

In general, all approaches to estimating costs of air pollution effects on materials are limited by the difficulty in quantifying the human response to damage based upon the ability and the incentive to pay additional costs (Yocom and Grappone, 1976). The physical damage approach requires specification of all relevant substitution possibilities and reliable exposure estimates. The welfare economic approaches, on the other hand, require control of several elements of survey bias, including "willingness to pay," and economic data that affect or clearly reflect choices.

10.5.2 Economic Loss Associated with Materials Damage and Soiling

Direct damage to materials is usually attributed to the corrosive action of SO_x, even though the interaction of SO_x and PM is recognized as an important determinant of the amount of actual damage (see Sections 10-2 and 10-3). Soiling of materials is attributed to PM.

10.5.2.1 Metal Corrosion and Other Damage to Materials Associated with Sulfur Oxides--Realistic estimates of the economic damage to metals from SO_x and PM must consider several factors, including avoidance costs, such as the costs of specific protective treatment. For metals, these costs include the use of anticorrosive primers, the practice of sandblasting before painting, and the use of acid-resistant paints.

Gillette (1975) reported significant reductions in economic damage to materials from SO_x due to improved air quality levels throughout the United States. Comparing annual SO_2 concentrations from more than 200 monitoring sites with the estimated inventory of materials exposed near these sites, he estimated that nationwide material damage decreased from more than $900 million in 1968 to less than $100 million in 1972, for a total of $1.4 billion in realized benefits over 5 years. These estimates were derived by distinguishing between physical and economic losses and by attributing current estimates of losses to current exposure levels.

The distinction between the physical and economic damage to materials is based upon the concept of normal or economic useful life of materials. Whereas physical deterioration of materials may occur at relatively low exposure levels, economic losses will occur only if the material requires early replacement or increased maintenance before its normal or economically useful life is spent. Gillette (1975) used Internal Revenue Service guidelines for service life assumptions, which may have introduced a downward bias to his damage estimates. Given the prevailing ambient concentrations observed, Gillette reported that most materials were not adversely affected economically except for metallic products that were subjected to corrosion or paint damage. While material losses were much greater during the early 1960's, the later losses were substantially lower and reflected the considerable improvement in air quality (Gillette, 1975).

A U.S. Department of Commerce report (Bennett et al., 1978) examined the cost of corrosion in the United States in 1975. Unfortunately, the developed damage costs were not pollutant-specific and were not associated with ambient concentrations of pollutants. The report estimated the total annual metallic corrosion cost at $82 billion with a model that incorporated a broad range of cost items (e.g., materials, labor, energy, and technical capabilities). About 40 percent of this cost, or $33 billion, was considered avoidable. Within this avoidable cost is, of course, the cost of air pollution, a portion of which is in turn the cost of metallic corrosion resulting from PM, SO_x, or both. The figure of $33 billion is thus only useful as an upper limit for present purposes since the estimate reflects economic damages from all pollutants as well as other avoidable costs.

Fink et al. (1971) estimated that corrosion of external metal structures caused by air pollution costs $1.45 billion annually in the United States, as shown in Table 10-9. As with the extensive Bennett et al. (1978) Department of Commerce report, these studies were not specific to single pollutants, nor were the damage costs associated directly with ambient pollutant concentrations. Furthermore, in some cases the cost estimates included material damage resulting from causes other than air pollution (e.g., the Fink et al. study included corrosion inside pipes of industrial systems). On the other hand, Haynie (1974) noted that within the estimate by Fink et al. of metal corrosion costs, damage to structural systems primarily constructed of galvanized steel accounted for more than 90 percent of the cost. No significant damage was assigned to corrosion of copper, aluminum, stainless steel, or lead. Haynie reasoned that, based on the data of Fink et al.(1971), the accelerated corrosion of zinc by SO_2 accounts for more than 90 percent of corrosion caused by air pollutants.

TABLE 10-9. SUMMATION OF ANNUAL EXTRA LOSSES DUE TO CORROSION DAMAGE BY AIR POLLUTION TO EXTERNAL METAL STRUCTURES FOR 1970

Steel system or structure	Basis for calculation	Annual loss in $1000
Steel storage tanks	Maintenance	$ 46,310
Highway and rail bridges	Maintenance	30,400
Power transformers	Maintenance	7,450
Street lighting fixtures	Maintenance	11,910
Outdoor metal work	Maintenance	914,015
Pole-line hardware	Replacement	161,000
Chain-link fencing	Maintenance and replacement	165,800
Galvanized wire and rope	Replacement	111,800
Transmission towers	Maintenance	1,480
		$1,450,165

Source: Fink et al. (1971).

The economic damage from corrosion can also be estimated by determining the annual cost of industrial paints used for corrosion control. This cost is estimated by summing the costs of surface preparation, paint, and labor. Coatings applied at this combined cost would protect metal structures against corrosion for about 10 years, whereas "ordinary" coatings of paint would provide protection for only 2 years. Moore and O'Leary (1975) investigated the painting of structural steel in bridges, which involves sandblasting the steel to produce a rust-free surface and to remove mill scale. Without such surface preparation, water is immediately absorbed and sets up a corrosion system, rusting occurs, and the paint surface deteriorates in 2 to 3 years. The metal surface is protected by a primer that inhibits rust formation, and the primer coat is covered with two coats of SO_2-resistant paint, such as vinyl resin, which is substantially more expensive than household paint. Banov (1973), Michelson and Tourin (1967), and others developed estimates to reflect these preparation and painting costs for protection of metal structures; however, sound fractional allocation of these costs to SO_x material damage is not available in the literature.

Stankunas et al. (1981) developed a geographically distributed inventory of exposed materials in the Boston metropolitan area. Isopleths for SO_2 were laid over a map reflecting exposed materials. Damage functions were applied to determine an estimate of SO_2 effects on materials for the entire area. Cost factors for repair were applied; the resulting annual cost estimate for SO_2 damage to paint in the Boston area was $31 million. For bare galvanized metal, the annual cost due to SO_2 was reported to be only $335,000. The authors noted that the amount of exposed bare galvanized metal and of structural concrete was low when compared to exposed painted surfaces. They also pointed out that the paint damage costs were derived using a damage function for oil-based house paint. However, its applicability is questionable due to the widespread shift to SO_2-resistant latex paints (see Table 10-5).

While this study represents a step toward developing and applying of realistic exposure data, a national estimate cannot be reliably extrapolated because of wide variations in environmental conditions (maritime versus inland locations, humidity levels) and material use (commercial versus industrial cities). Similar studies of cities in various regions may provide a basis for a more reasonable composite national estimate.

Damage from pollutants during the manufacture of electrical components must be controlled even in clean environments. To prevent such problems, parts are fabricated in "clean rooms" with filtered air. Equipment malfunctions in plant or service areas lead to additional maintenance costs for cleaning, repairing, or replacing defective equipment. ITT (1971) estimated $15.5 million per year in added costs for clean rooms and maintenance. Robbins (1970) conducted a survey of the effects of SO_2 and particles on electrical contacts such as those in switches, relays, connectors, and computers. To reduce corrosion, contacts are electroplated with corrosion-resistant metals (e.g., gold, platinum, palladium, and silver). Less expensive metals are susceptible to corrosion failure, mostly from the action of SO_2 and H_2S. Robbins estimated that 15 percent of the gold and platinum used in the United States for electrical contacts in 1970 was used specifically to combat SO_2 corrosion, with the remainder for protection against other environmental pollutants. Protection against all deleterious environmental conditions, however, is routinely provided in clean environments due to high replacement costs associated with electrical system downtime. Thus, costs for protection of electrical hardware cannot be allocated directly to ambient SO_x, PM, or both.

10.5.2.2 Soiling of Paint and Other Materials Associated with Particulate Matter--Studies of soiling have historically focused on cleaning costs, with the assumption that as particulate pollution increases, so does the frequency of cleaning and maintenance. This was the thesis of one of the first national estimates of soiling costs associated with air pollution. The Beaver Report (1954) suggested an annual total for damage by all forms of air pollution in Great Britain of 152 million pounds sterling in direct costs, of which 25 million was for laundry, 30 million for painting and decorating, and 20 million for cleaning and depreciation of buildings other than houses; thus, about half the total imputed cost of pollution was attributed to soiling associated with particulate pollution.

In the United States, the same hypotheses underlay approaches to quantify soiling costs associated with particulate pollution, or, as measured in the United States, TSP. Almost without exception, these studies have focused on household cleaning and maintenance. Michelson and Tourin (1966) compared cleaning and maintenance costs in Steubenville, Ohio, where the average TSP level was 383 $\mu g/m^3$ with costs in Uniontown, Pennsylvania, where TSP averaged 115 $\mu g/m^3$. The study was based on a 30-percent response to a questionnaire mailed to 2 to 6 percent of the households in these communities. Michelson and Tourin reported that per capita costs for cleaning and maintenance were $84 higher in Steubenville than in Uniontown. However, the results can be questioned on the following grounds: failure to account for socioeconomic factors, inadequate tests of statistical significance, and consumer attitudes.

In a second study using a similar approach, Michelson and Tourin (1967) sampled communities in the Washington, D.C. area (Suitland and Rockville, Maryland and Fairfax, Virginia). The TSP average levels for the three communities were 85, 68, and 60 $\mu g/m^3$, respectively. Though the study did show an increase in frequency of cleaning with increased TSP levels, errors associated with measurement techniques, averaging over a community, and influence of socioeconomic factors could be of the order of the small differences reported.

Ridker (1967) undertook two studies attempting to identify and quantify the soiling costs of air pollution. In 1965 Ridker divided Philadelphia into high, medium, and low pollution zones, and administered a series of questions to residents of individual housing units selected at random in each zone. Ridker diverged from previous questionnaires administered in other studies by correcting for attitudes toward cleanliness, and asking willingness-to-pay questions on controlling or mitigating air pollution soiling. The results of the study, however, were largely inconclusive.

Ridker (1967) also conducted a study on an air pollution episode in Syracuse, New York. A survey instrument and selection of a sample, designed to collect data within a very short time after the pollution incident, were used. The results of the household survey were better than the one conducted in Philadelphia. However, Ridker felt that the difference in response to questions concerning clean-up after the pollution episode could be explained by variables other than pollution. Respecification of the estimating equation to obtain a cost curve independent of the effect of the intervening variables (by combining questionnaire responses to intervening variables with a pollution index) gave an upper and lower bound to the estimated damages. Nevertheless, the results of the Syracuse study are limited to episode-type situations and cannot be generalized.

In 1968, the National Air Pollution Control Administration (NAPCA) became concerned that the previous studies, individually and as a group, lacked statistical validity to form a basis for assessing the benefits of air pollution control. To overcome the shortcomings of the previous studies, NAPCA commissioned the Booz, Allen and Hamilton (1970) (BAH) study. The objective of the BAH study was to determine residential soiling costs of particulate air pollution for the 11-county Philadelphia area, stretching approximately from Wilmington, Delaware, to Trenton, New Jersey. Although the primary purpose of the study was to determine the residential soiling costs in the 11 county area, it also had the alternative objectives of providing methods of estimating residential soiling costs under various abatement strategies, comparing its findings with previous work (Michelson and Tourin), and developing a sampling methodology which could be applied in other metropolitan areas.

Important to note is that the BAH study was the first of its kind to combine a well-designed sampling strategy and questionnaire. Although limited to the Philadelphia region, it continues to serve as the only empirical basis for estimating soiling costs not only for the Philadelphia region, but also the entire nation.

The BAH questionnaire and interview procedure consisted of eliciting information on the following:

(1) Attitudes toward household cleaning and maintenance (e.g., neat and tidy, lackadaisical, etc.).

(2) Demographic and economic information, including but not limited to educational level and occupation of household, number of children living at home, marital status, total family income, national origin, etc. (See Table 10-10.)

(3) Maintenance and cleaning performance information on 27 indoor and outdoor tasks. (See Table 10-11.)

(4) Cost of maintenance and cleaning operation estimates, including whether the task was performed by "do it yourself" method or a contractor, cost of hired labor, cost of materials, etc.

TABLE 10-10. SELECTED CHARACTERISTICS OF HOUSEHOLDS IN FOUR AIR POLLUTION ZONES
(Booz, Allen and Hamilton, 1970)

		Weighted Total	Zone			
			1	2	3	4
All Households	Number* -	---	471	421	299	251
	Percent -	100%	100%	100%	100%	100%
Race						
White		79	93	78	57	70
Non-White		21	7	22	43	30
Family Income of Household						
Under $6,000 annually		32	21	32	45	49
$6,000 - $9,000 annually		29	27	25	36	32
$10,000 or more annually		31	40	34	16	13
Marital Status of Household Head						
Married with minor children at home		65	69	61	61	65
Married, no minor children at home		30	27	33	34	28
Not married		5	4	6	5	7
Tenure of Household						
Own home		74	79	74	67	64
Rent home		26	21	26	33	36
Dwelling Units in Structure						
Single-family structure		51	62	42	44	43
Multifamily structure		43	33	51	52	47
Outside Wall Surface Areas						
More than 10% painted wood		12	16	15	6	3
More than 10% aluminum siding		8	10	8	4	3
More than 75%.brick or masonry		45	31	40	59	79
Number of Rooms in Household**						
10 or fewer		36	27	33	45	59
More than 10		64	73	67	55	41
Mean Number of People Per Household		3.54	3.63	3.47	3.40	3.54

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* Some categories may not total to 100% because of nonresponses and/or multiple responses.

** Closets and bathrooms/lavatories included in count.

TABLE 10-11. 27 CLEANING AND MAINTENANCE OPERATIONS SEPARATED BY SENSITIVITY TO AIR PARTICULATE LEVELS IN FOUR POLLUTION ZONES
(Booz, Allen, and Hamilton, 1970)

OPERATION/TASKS	ZONE 1		ZONE 2		ZONE 3		ZONE 4	
	Mean Annual Frequency	% Households Performing	Mean Annual Frequency	% Households Performing	Mean Annual Frequency	% Households Performing	Mean Manual Frequency	% Households Performing
(Number of Households Sampled)	(471)		(421)		(299)		(251)	
Sensitive to Air Particulate Level								
Inside								
1. Replace Air-Conditioner Filter	0.36	20.4	0.50	26.1	0.30	17.1	0.98	21.9
2. Wash Floor Surfaces	40.55	96.0	42.06	96.9	42.74	94.3	45.17	97.2
3. Wax Floor Surfaces	13.76	65.2	14.36	55.6	16.80	56.5	13.01	51.8
4. Wash Windows (inside)	10.06	97.2	11.78	97.9	12.74	97.0	18.95	98.0
5. Clean Venetian Blinds/Shades	4.04	34.0	6.17	48.2	9.13	62.5	9.21	59.4
Outside								
1. Clean/Repair Screens	0.80	56.5	0.93	51.1	0.79	45.2	1.50	49.4
2. Wash Windows (outside)	4.25	89.5	4.59	89.5	6.17	86.3	10.09	89.3
3. Clean/Repair Storm Windows	2.07	55.6	1.60	48.5	2.12	47.8	3.69	41.0
4. Clean Outdoor Furniture	2.50	31.2	4.29	29.7	3.52	22.7	1.19	6.4
5. Maintain Driveways, Walks	3.98	35.2	7.25	32.8	7.33	27.8	6.84	21.1
6. Clean Gutters	1.12	45.9	1.54	41.8	1.35	21.7	2.80	19.5
Not Sensitive to Air Particulate Level								
Inside								
1. Painting Walls/Ceilings	0.30	76.6	0.38	72.0	0.54	67.9	0.29	59.0
2. Wallpapering	0.07	36.9	0.10	45.4	0.12	43.1	0.12	52.6
3. Washing Walls	3.04	42.7	2.68	41.3	2.99	45.8	3.36	37.1
4. Replacing Furnace Filter	0.98	50.1	0.84	38.7	0.63	31.4	0.42	25.5
Cleaning/Oiling Air-Conditioners	0.25	12.3	0.40	21.4	0.50	15.1	0.32	16.7
Cleaning Furnace	0.74	62.4	0.77	51.8	0.38	40.5	0.46	46.6
Dry-Cleaning Draperies	0.28	20.2	0.25	23.0	0.27	21.1	0.51	17.5
Dry-Cleaning Carpeting	0.43	12.5	0.11	13.5	0.09	10.7	0.07	6.8
Shampooing Carpeting	1.43	47.8	1.26	44.2	1.13	37.8	1.23	39.0
Shampooing Furniture	0.62	12.5	0.64	14.7	0.71	13.4	0.30	12.0
ide								
Painting Outside Walls	0.11	38.6	0.10	28.5	0.04	11.4	0.03	10.4
Painting Outside Trim	0.22	66.5	0.29	68.9	0.20	54.5	0.20	55.0
Cleaning/Repairing Awnings	0.05	4.0	0.05	4.3	0.04	4.3	0.31	5.2
Washing Automobiles	1[illegible]8	80.9	16.63	66.0	14.94	50.8	12.59	49.8
Waxing Automobiles	3.47	44.4	3.49	42.5	2.70	29.8	1.75	28.3
Maintaining Shrubs, Flowers, etc.	15.47	71.3	13.16	64.6	10.15	38.8	3.27	14.3

(5) Residence characteristics including length of tenure of occupants at residence, home rented or owned, size and number of rooms, type of material used in inside and outside of structure, heating and air conditioning characteristics, fuel type, number of windows, etc. (See Table 10-10.)

The BAH study covered four pollution zones in the Philadelphia area defined by ambient air monitoring data. The four zones were identified as: (Zone 1) less than 75 $\mu g/m^3$, (Zone 2) 75 to 100 $\mu g/m^3$, (Zone 3) 100 to 125 $\mu g/m^3$, and (Zone 4) greater than 125 $\mu g/m^3$. Frequency rates of performing 27 cleaning and maintenance tasks (15 inside, 12 outside) were determined for each pollution zone by various demographic and attitudinal subgroups (see Tables 10-11 and 10-10). The BAH study used appropriate statistical survey techniques (stratified random sampling) and collected the best set of frequency of performance and cost data to date. The overall finding by the BAH researchers was:

> that over the 27 cleaning and maintenance operations studied the range of annual air particulate levels experienced in the Philadelphia area (approximately 50 to 150 micrograms per cubic meter) has no measurable effect on out-of-pocket cleaning and maintenance costs for the residence of the over 1,500,000 house holds in the area.

A small number of contractors were interviewed in deriving the cost information. However, not only were no statistical sampling procedures used in their selection, but also the estimates of costs failed to consider size of the project or types and conditions of materials to be restored. It is not unexpected, therefore, that no significant relationship was found between the prices for any given service and the location in the four pollution zones of the places of business of the contractors (from which labor costs were estimated in BAH) within the region. All estimated costs were applied uniformly across the region.

The researchers stated other conclusions:

(1) Activities related to "looking outdoors" (e.g., washing windows) were more associated with air pollution levels. These tasks represented nonsignificant low "out-of-pocket costs" because few households contracted out such types of work.

(2) Residents of high pollution zones believed the effect of pollution on cleaning costs to be greater than the residents of low pollution zones.

The BAH findings for Philadelphia should not be entirely accepted without at least considering that the value of the direct personal labor or time of a "do-it-yourself" resident was not included as a cost.

Although the methodological approach employed by BAH is applicable to other sampling sites, the specific findings of the Philadelphia study cannot be automatically assumed valid for other metropolitan areas of the nation. Factors affecting cleaning costs, such as hourly rates and unit costs of contractor work, may differ from one major region to another. Different types of particulate matter found from region to region produce different kinds and degrees of soiling. Also, Philadelphia, as well as many other urban regions of the Eastern U.S., contains many dwellings constructed of brick or masonry. Cleaning estimates based upon Philadelphia, which contains up to 80% brick or masonry structures in the most polluted zone, cannot be extrapolated to other regions in the nation where use of other types of construction materials may prevail. A need clearly exists for studies similar in approach to the BAH study to be conducted in other regions of the nation.

In 1972, Spence and Haynie presented a survey and economic assessment of the deterioration and soiling of exterior paints caused by air pollution. Although the authors discussed effects of SO_2 and other pollutants on paint, their economic assessment reflects only the effects of soiling by PM. Of an estimated total annual economic loss of \$704 million in 1968 dollars, \$540 million was for increased exterior household painting. No damage functions were available for use in the estimate. Instead, the calculation was based on the assumption that exterior household paint service life is reduced by half in an (urban) area averaging 110 $\mu g/m^3$ TSP from a service life of 6 years in a (rural) area averaging 40 $\mu g/m^3$. This length of service life was in turn based on frequency of painting. Spence and Haynie explained the derivation of the service life assumption as follows: "The Michelson linear relationship... predicts service life of 8 and 2 years, respectively, for paint exposed to 40 and 110 $\mu g/m^3$ of particulate matter. The Booz, Allen and Hamilton, Inc. study shows no difference in service life for paint exposed over this concentration range, the average paint life being 3.14 years. Because neither study was complete in its analysis, paint lives are estimated at 6 and 3 years, respectively, for rural and urban areas."

Narayanan and Lancaster (1973) conducted a questionnaire survey in a rural area and a polluted area in New South Wales, Australia, to determine the difference in cost of household upkeep. The cost of maintaining a house in the Mayfield area was about \$90 per year higher than in the relatively unpolluted Rotar area. The authors attributed this cost differential to the higher levels of air pollution and airborne particulate matter in Mayfield. However, the accuracy of the cost data was considered questionable, because like the earlier Michelson and Tourin studies, socioeconomic factors, including respondents' attitude, were not dealt with and could effect a bias in the estimates.

A study by Liu and Yu (1976) was designed to generate physical and economic damage functions, by receptor, for SO_2 and TSP, and to establish cost/benefit relationships. The study included the effects of air pollution damage on health, materials, vegetation, and household soiling. These authors used the BAH results on cleaning frequency for nine household cleaning tasks, and the data for TSP levels, in a Monte Carlo technique. The technique created a sample of data pairs (cleaning frequency and TSP level) for each cleaning task. The authors regressed cleaning frequency on TSP levels without controlling for the social and economic factors reported and included in the BAH report. Increased cleaning frequencies were transformed into monetary units for each task and were used to calculate decreases in cleaning costs as TSP is lowered to 45 $\mu g/m^3$. These costs were extrapolated to 148 SMSA's, then summed for a national estimate of $5 billion household soiling cost attributed to TSP. Liu and Yu have been criticized for their disregard of the socioeconomic factors in the BAH data base and for ignoring evidence regarding the insensitivity of high-cost cleaning and maintenance tasks to TSP levels.

Watson and Jaksch (1978) returned to the Booz, Allen and Hamilton (1970)(BAH) study for their assessment of benefits associated with decreased TSP levels. They introduced a significant extension of the earlier work by including in the benefits of pollution control the psychological and other advantages of living in a cleaner environment. Watson and Jaksch estimated the value of these psychological and health benefits by applying the standard measure of net contribution to consumer welfare, namely the excess of the sum that consumers would be willing to pay for the attained level of cleanliness over the actual cost to them of attaining that level.

To estimate the cost of achieving a given level of cleanliness and the effect on that cost of reducing the concentration of particulate matter, they relied largely on the work of Beloin and Haynie (see Table 10-8). The formulas derived by Beloin and Haynie lead directly to estimates of the cost of maintaining a given level of reflectance of different surface materials, which, however, is not the same thing as the perceived level of cleanliness, though related to it. "Cleanliness" is a vague concept, and in order to give it the precision necessary for estimation, Watson and Jaksch posited that it is the reciprocal of the difference between the actual reflectance of a surface and its maximum reflectance, raised to a power that depends on the rate at which reflectance decreases over time. This enabled them to express the marginal cost (or price to the consumer) of maintaining a given average level of cleanliness, Q, in the form:

$$MC = aP^{\alpha}Q, \qquad (10\text{-}14)$$

where a and α are empirical constants and P denotes the ambient concentration of particulate matter. Beloin and Haynie's data for only one kind of surface indicate a value of α of approximately 0.5. Less detailed observations by Esmen (1973) are consistent with $\alpha \cong 2$. This range of estimates was used in the subsequent analysis. It should be noted that this formula depends on both the empirical studies of reflectance and the assumed relation between reflectance and perceived cleanliness.

In the presence of a given ambient concentration of particulates, consumers can be expected to choose the level of cleanliness at which the cost of further improvement, as revealed by equation (1), is just equal to the amount they are willing to pay for it. In order to estimate the amounts that consumers are willing to pay for increases in cleanliness, Watson and Jaksch derived a demand curve for cleanliness by a detailed reanalysis of the findings of the Booz, Allen and Hamilton study of the frequency of household cleaning in Philadelphia. As previously mentioned, that study indicated that the ambient concentration of pollutants had no significant effect on the frequency with which cleaning tasks were performed. Thus, people put up with lower levels of cleanliness in heavily polluted areas (where maintaining given levels is expensive) than in less polluted areas. In fact, to the extent that the cost of maintaining a given level is proportional to the frequency of cleaning required, households adapted to levels that cost the same amount to attain at all levels of ambient pollution. That observed behavior is consistent with a demand curve with constant elasticity of -1. Accordingly, that type of demand curve was adopted. It implies that out-of-pocket costs of home maintenance are not affected by ambient particulate levels, while psychological satisfactions are affected.

Watson and Jaksch then brought the estimated cost and demand curves together in the usual manner to estimate the changes in consumer welfare associated with changes in ambient concentration, by calculating the level of cleanliness that would be chosen as a function of the ambient concentration and then the area under the demand curve up to that level, less the cost of attaining that level. The changes in consumer welfare corresponding to changes in particulate matter concentrations were estimated by comparing the results of that computation for different ambient concentrations.

These estimates were made for the households in the BAH survey, then inflated to cover the entire Philadelphia metropolitan area, and finally extended to 123 SMSA's in the United States, with allowance for the differing prevailing TSP concentrations in the different SMSA's. The results are summarized in Table 10-12. Converted to 1978 dollars, the nationwide gains to consumers from attaining primary TSP standards in all SMSA's are valued at from $1.4 billion per year to $5.1 billion, depending on which value of alpha (0.56 or 2.0) is used. The corresponding estimates for achieving the secondary standard are $2.4 to $9.1 billion. No allowance is made in this table for any sources of uncertainty or estimating error other than the range of assumed values of alpha (see Figure 10-10).

The Watson and Jaksch (1982) study is conceptually sound. The data and statistical methods on which the study rests, however, limit full acceptance of its generalized inferences. Application of the method with better defined parameters and to data sets outside the Philadelphia region is necessary. Nevertheless, despite these limitations, the study remains a primary source of estimates of the benefits to consumers of the reductions in soiling achieved by reducing ambient TSP concentrations.

A study by Hamilton (1979) used the framework developed by Watson and Jaksch to estimate soiling benefits for six California SMSA's. Hamilton assumed a unit elasticity of demand for cleanliness. He assumed a 25 percent reduction in total suspended particulate matter, which produced estimated benefits of $40 per household, which is comparable to the estimates derived

TABLE 10-12. ANNUAL WELFARE GAIN FROM ACHIEVING PRIMARY AND SECONDARY STANDARDS FOR TSP CONCENTRATION

(1971 Prices)

	Primary Standard	Secondary Standard
Average gain per household, Philadelphia sample, dollars	\$16 - 57	\$30 - 112
Aggregate gain, Philadelphia \$ million	23 - 85	44 - 165
Aggregate gain, 123 SMSA's, \$ million	860 - 3200	1500 - 5700

Source: Watson and Jaksch (1982).

Note: Gains calculated from reductions from TSP concentrations in 1970. Primary standard: 75 $\mu g/m^3$, annual average. Secondary standard: 60 $\mu g/m^3$. Ranges correspond to $\alpha = 0.56$ (lower limit) to $\alpha = 2$ (upper limit).

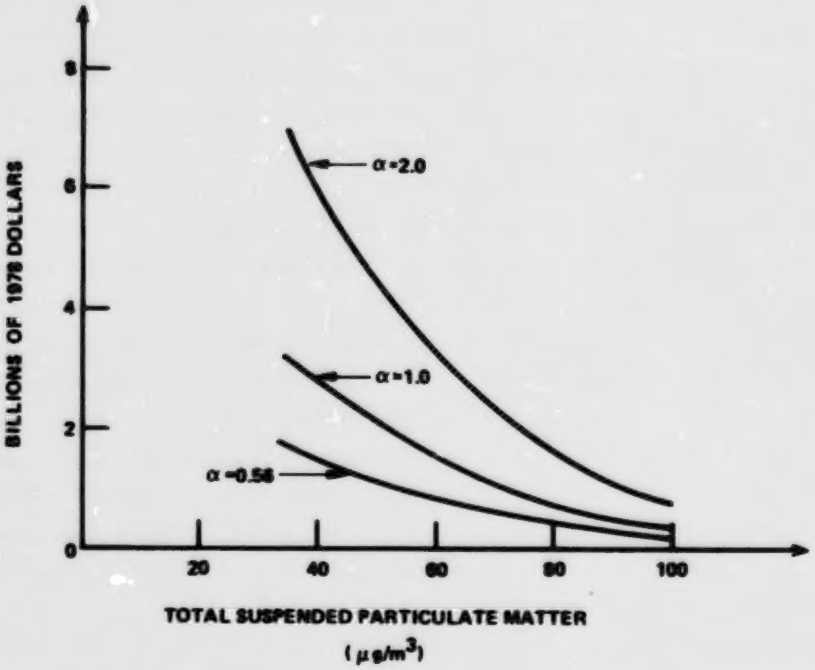


Figure 10-10. Increases in particulate matter concentrations are plotted against reductions in outdoor cleaning task benefits (1978 dollars). The range of benefits increases progressively as pollution is reduced.

Source: Derived from Watson and Jaksch (1982).

by Watson and Jaksch. Total benefits for a 25 percent reduction in PM for the six California SMSA's (including Fresno, Los Angeles-Long Beach, Sacramento, San Bernadino-Riverside-Ontario, San Diego, and San Francisco-Oakland) are $223 million, in 1978 dollars.

10.5.2.3 Combined Studies--Salmon (1970) was one of the first to provide a cost estimate of soiling and material damage associated with both ambient PM and SO_2 concentrations, shown in Table 10-13. He calculated economic loss by determining the value of materials exposed to pollution and then multiplying by the estimated difference in useful lifetime between clean-rural and polluted-urban areas. The value of exposed materials was derived by multiplying annual production by a product lifetime estimate and then applying a labor-factor estimate and an exposure-factor estimate. Salmon cited economic damage from SO_x in the United States to the following materials, listed in decreasing order of the extent of damage: metals, cotton, finishes and other coatings, building stone, paints, paper, and leather. Deterioration of paint was estimated at $1.2 billion, soiling of paints at $35 billion. Soiling costs attributable to PM were calculated to be $99 billion. In discussing limitations to his study results, Salmon pointed out that data were available to examine the economic value of materials loss for only six material categories: stainless steel, zinc, building stone, leather and paper, cotton, and paint. He cautioned that his cost estimate indicated susceptibility to economic loss or potential loss and should not be interpreted as actually incurred economic loss. He claimed only that the study "may be a reasonably good approximation of reality." The ranking of materials by decreasing order of potential economic loss may be reasonably used for setting research priorities, which was the major purpose of the study. As Table 10-14 shows, however, the Salmon study has been used quantitatively to some extent in nearly every national estimate for materials damage costs attributable to air pollution done since 1970. In general, his figures are relied upon for costs not estimated elsewhere, reflected in the "other" category of materials in Table 10-14. (Discussion of Waddell (1974), Yocom and Grappone (1976), and Freeman (1979b) later in this section will show how each of these studies used the Salmon estimates, directly or indirectly.) Nevertheless, no subsequent study has arrived at costs nearly as large as Salmon's.

Waddell (1974) used Salmon's list of 32 economically important materials significantly affected by air pollutants and associated damage costs, revising estimates for materials about which more recent reports had become available. For those costs associated with paint, he substituted the Spence and Haynie (1972) estimate of $704 million. Gillette's SO_2 damage estimate as developed in an earlier 1973 version was used to replace the SO_2 damages assigned to zinc, carbon and alloy steel, fibers, cement and concrete, building bricks and stone, plastics, paper, leather, and wool. Gillette's estimate of damage to these materials for 1970 attributable to SO_2 was $400 million. It should be noted that the Gillette (1975) study focused on damage to metals and paints and considered damages to other materials to be insignificant. Waddell assigned a zero value to copper, aluminum, stainless steel, and lead, based on the conclusions of Fink et al. (1971). Waddell then totaled the costs associated with

Table 10-13 ECONOMIC LOSS, MATERIALS DAMAGE ATTRIBUTED TO AMBIENT EXPOSURE TO SO_x AND PM, ESTIMATED BY SALMON (1970) (IN BILLIONS OF 1970 DOLLARS)

Material	SO_x	PM	Total
Paint	1.195	35.0	36.2
Zinc	0.778	24.0	24.8
[a] Fibers	0.358	0.5	0.9
Cement and concrete	0.316	5.4	5.7
Nickel	0.260	1.0	1.3
Tin	0.144	0.1	0.2
Aluminum	0.114	4.9	5.0
Copper	0.110	0.2	0.3
Carbon steel	0.054	<0.1	0.1
Building brick	0.024	0.1	0.1
Paper	0.023	1.1	1.1
Leather	0.021	2.5	2.5
Glass	<0.001	19.0	19.0
Building stone	0.018	0.1	0.1
Wood	0.018	<0.1	0.1
Brass and bronze	0.014	0.2	0.2
Magnesium	0.013	<0.1	<0.1
Alloy steel	0.009	<0.1	<0.1
Bituminous materials	0.002	<0.1	<0.1
Gray iron	0.002	<0.1	<0.1
Stainless steel	0.002	<0.1	<0.1
Clay pipe	0.001	<0.1	<0.1
Chromium	0.001	0.2	0.2
Malleable iron	0.001	<0.1	<0.1
Silver	0.001	<0.1	<0.1
Gold	0.001	<0.1	<0.1
Plastics	<0.001	4.7	4.7
Lead	<0.001	0.1	0.1
Molybdenum	<0.001	<0.1	<0.1
Rubber	<0.001	0.1	0.1
Refractory ceramics	<0.001	<0.1	<0.1
Carbon and graphite	<0.001	<0.1	<0.1
TOTALS[b]	3.8	99.2	102.7

[a] Combined effects of SO_x, O_3 and NO_x on cotton ($152 million), wool ($99 million) nylon ($38 million) and other synthetics ($69 million)

[b] Not additive, due to rounding.

Source: Salmon (1970).

TABLE 10-14. ESTIMATES OF MATERIALS DAMAGE ATTRIBUTED TO SO_x AND PM IN 1970 (IN MILLIONS OF 1970 DOLLARS)

	ESTIMATES							
	WADDELL (1974)			YOCOM AND GRAPPONE (1976)			FREEMAN (1979b)	
MATERIAL CATEGORY	SO_x	PM	TOTAL	SO_x	PM	SO_x/PM Total	SO_x/PM[a]	SOURCES
PAINTS	100	100	200	200	500	700	704	Spence and Haynie, 1972
TEXTILES AND DYES	--	--		--	--	--	636	Salvin, 1970
METAL CORROSION	400	--	400	400	--	400	400	Gillette, 1975[b]
ELECTRICAL SWITCHES AND COMPONENTS	included in metal corrosion			included in metal corrosion			80	Robbins, 1970 and ITT, 1971
OTHER[c]	100	200	300	300	100	400	400	Salmon, 1970
TOTAL	600	300	900	900	600	1,500	2,200	

[a] No allocation of cost to PM or SO_x specifically.

[b] Nickel, tin, brass, bronze, magnesium, gray iron, malleable iron, chromium, molybdenum, silver, gold, clay pipe, glass, refractory ceramics, carbon, and graphite.

[c] Waddell used earlier (1973) version of Gillette (1975) report.

materials remaining on Salmon's list and assigned the resulting $400 million collectively to the category "other." No evidence exists, however, for a specific association of SO_x or PM with effects on these materials. In allocating these costs specifically to PM or SO_x, Waddell made an adjustment to the $704 million paint damage estimate. He considered that the portion of the costs attributable to soiling of household paint by PM ($500 million) was included in the soiling estimate. The remaining $200 million was evenly divided between PM and SO_x. The "other" category (see Table 10-14), derived from Salmon (1970) was allocated in proportion to criteria pollutant emissions, excluding carbon monoxide. On this basis, $100 million was assigned to SO_x and $100 million to PM for effects on materials not addressed by Gillette (1975) or Spence-Haynie (1972) or excluded by Fink (1971).

In developing a national estimate for soiling damages attributable to particulate pollution, Waddell (1974) concluded that, on the basis of studies published up to 1973 designed to determine household soiling costs attributable to PM, "the evidence to date indicates air pollution does not have significant economic effects in terms of household maintenance and cleaning operations." He declined to make an estimate of economic damages associated with soiling on the basis of these studies.

As an alternative, Waddell turned to studies relating residential property value differentials to air pollution. *Of the nine studies Waddell reviewed, six included PM and SO_2, while three related property value differentials only to SO_2 (as measured by sulfation). Of the six that included PM, three studies found significant negative correlations between property value differentials and particulate pollution levels. The strongest associations related property value differentials to sulfation rates. From these studies, Waddell calculated "marginal capitalized sulfation damage coefficients" for residential structures of $200-$500 for 0.1 mg/ SO_3/100 cm^2/day. Using these coefficients, he estimated the annual national cost for 1970 of air pollution damage measured via the property differential method to be $3.4 to $8.4 billion. He used the midpoint of this range, $5.9 billion, as a best annual estimate. In estimating and allocating gross damage estimates between pollutants, Waddell adjusted the $5.9 billion estimate by subtracting $50 million (1970 dollars) for ornamental losses as determined by Benedict, et al., (1971). By assumption, Waddell postulated the property value estimate to measure aesthetic and soiling costs. He then assumed that the total cost of $5.8 billion* could be allocated by evenly *dividing the damage between PM and* SO_x, or $2.9 billion each.

Yocom and Grappone (1976) also estimated economic costs for air pollution damage to materials, in what was essentially a revision of Waddell's study. They concluded that the cost of materials damage attributable to SO_x and PM in 1970 was $1.5 billion. As may be seen in Table 10-14, they departed from Waddell's approach in two categories: (1) The entire "other" category derived from Salmon (1970) was assigned to PM and SO_x (both Salmon and Waddell had considered these costs attributable to all major air pollutants); and (2) The paint cost was not adjusted to distinguish between industrial and exterior house paint. These two differences account for the $0.5 billion difference between the Waddell (1974) and Yocom and Grappone (1976) estimates. The latter made no attempt to calculate a separate estimate for soiling costs. Considering that the household paint soiling figure would have been assigned to a soiling estimate had Yocom and Grappone developed one, the two studies are essentially in agreement.

Freeman (1979b) estimated benefits from reduced materials damage and soiling associated with improvements in air quality from 1970 to 1978. He assumed a 20 percent improvement in levels of SO_2 and in levels of TSP. To establish a basis for evaluation of the realized benefits associated with materials damage, Freeman essentially revised Waddell's (1974) estimate and updated the result to 1978 dollars. The significant features of the revision, as shown in Table 10-14, are that Freeman included a $636 million figure for damage to textiles and dyes, carried forward the entire $400 million "other" category derived from Salmon (1970), did not assign the $500 million related to soiling of household paint to his separate soiling estimate, and included a figure for damage to electrical contacts and switches, which Waddell had assumed to be included in metal corrosion. Attribution of $636 million in textile and dye damage to PM and SO_x was based on a study by Salvin (1970). Later evidence, as discussed in Section 10.2.3, has shown effects of SO_2 on dyes to be insignificant compared to the effects of other pollutants, notably NO_x, although some evidence exists for SO_2 effects on fabric strength. Similarly, the damage to electrical contacts and switches does not reflect recent practice regarding use of corrosion-resistant metals. Freeman's estimate of damage to materials was not broken out for SO_x and PM separately; his estimate for damages attributable to both pollutants in 1970 was $2.2 billion, equivalent to $3.7 (±1.9) billion in 1978 dollars. Freeman then calculated the benefits for a 20-percent improvement in SO_x and TSP levels from 1970 to 1978; realized benefits were estimated to be $0.7 (±0.4) billion per year.

In calculating benefits for reduced soiling and cleaning attributable to particulate pollution, Freeman (1979b) also used the work of Watson and Jaksch (1978) to arrive at an estimated range of $1.5 to 11.7 billion in 1978 dollars. Based upon 1970 and 1978 national ambient particulate matter measurements, Freeman (1979b) inferred that between a third and a half of the Watson and Jaksch (1978) estimate of the benefits of attaining the secondary standard had been realized by 1978. Freeman's calculations showed this to be a range between $0.5 and $5.0 billion per year, or a reasonable midpoint estimate of about 2.0 billion per year. Incremental benefits of moving from 1970 particulate matter levels to achievement of the secondary standard in 1978 are estimated at $2.6 billion per year. However, Watson & Jaksch revised their paper subsequent to the publication of Freeman (1979b). The later Watson & Jaksch (1982) paper, therefore, will result in Freeman's estimates being changed.

In a report prepared for the National Commission on Air Quality, SRI International (1981) developed estimates of nonhealth benefits of meeting the secondary national ambient air quality standards. To estimate benefits resulting from reduced material damage, SRI used Salmon's (1970) estimate of pollution damage for their base year of 1969. More recent physical damage functions were incorporated when available, and a national stock inventory for 1979 was developed. Salmon's assumptions regarding economic life of materials was also employed in the SRI estimate. In calculating benefits associated with materials damage reduction from 1969 to 1979, SRI assumed no significant change for TSP and a 20 percent improvement in SO_2 levels. The resulting estimate of benefits associated with meeting the secondary standards was $3.9 billion in 1980 dollars. The authors stated that the estimate was similar to the conclusion reached by Freeman (1979b), although the two studies were not directly comparable.

In considering the benefit of reduced soiling, SRI reviewed the state of the art of the economics associated with soiling and concluded that: "(i) we are unable to quantify household soiling effects from alternative particulate levels; (ii) there is no basis at this time for determining indoor-outdoor pollution concentration ratios and (iii) the soiling effects of large versus small particulates [sic] are not well understood." To provide a rough estimate of soiling-related benefits, SRI applied a $6.63 marginal benefit of soiling per household per unit change in TSP concentrations to counties violating the secondary TSP standard to arrive at an estimated $0.5 billion benefit for meeting the secondary standard. The marginal benefit was derived from a damage function appearing in an unpublished EPA report. The very preliminary basis for the SRI soiling estimate diminishes its immediate utility; however, the identification by SRI of problems and research needs in the area of soiling is useful in evaluating existing estimates of soiling benefits.

Freeman (1979b) noted that in the materials damage area, "the weakest link in most of these (materials damage cost) studies is the estimation of quantities of material at risk and exposure levels." While further application of the exposure inventory (Stankunas et al. 1981) approach may provide information necessary for more reliable application of the physical damage function approach, new approaches to estimating the cost of pollutant effects on welfare, including materials damage, have been recently developed.

Manuel et al. (1981) developed an estimate of benefits to be realized from reductions in ambient SO_2 and TSP concentrations. They drew heavily on econometric analysis to identify the influence of changes in TSP and SO_2 levels on certain production and consumption decisions of selected components of the agricultural, manufacturing, household, and electric utility sectors of the economy. The approach allows consideration of substitution possibilities by both buyers and suppliers. Indices of decision behavior of decision-making units within each sector were chosen to reflect the effects of TSP and SO_2 on agricultural yield, materials damage, and soiling. The theoretical approach and empirical methods employed appear to be well based. However, additional analyses of the study results are needed to explain more fully certain implied behavioral adjustments. In addition, more extensive air quality data and more detailed economic data would have permitted not only greater coverage but might have also served to reduce the degree of uncertainty associated with the estimated benefits.

10.5.3 Estimating Benefits from Air Quality Improvement, 1970-1978

Although both the data base and the method used for estimating benefits associated with reduced materials damage and soiling are fraught with shortcomings, it is possible to judge the direction and rough order of magnitude of changes in these benefits as a function of changes in air quality. Certain features of the data base discussed in Section 10.5.2 must be carefully considered.

Between 1970 and 1978, almost without exception, aggregate estimates of materials damage and soiling used 1970 as the base year for comparison with later years. Major studies of material damage performed after 1970 relied principally on the physical damage function approach and sought to revise or update relevant portions of cost estimates of 1970 damage. As a result, the data base remained essentially unchanged. Largely because of this, estimates of materials damage costs vary within a much narrower range than do estimates of soiling costs, where a variety of approaches have been applied. These approaches were used to develop estimates for a specific area or region of the country, then extrapolated for a national estimate. Both differences in approach and extrapolation partially account for the wide range in resulting estimates of aggregate soiling costs.

Most cost estimates are based on PM or SO_x levels over and above some base level. Because of changes in analytical techniques and in number and location of monitoring sites, determination of reduction in SO_2 levels between 1970 and 1978 is difficult and should be approached with caution. Trends for TSP are less equivocal because, although monitoring sites were changed, the methodology remained the same (see Chapters 3 and 5). Cost (benefit) estimates of increasing (decreasing) pollutant levels must include any uncertainties of the air quality trends data. For the purposes of this section, SO_2 and TSP trends discussed in Chapter 5 will be used.

Estimates of cost associated with SO_x are dominated by damage to materials, principally metals, exposed outdoors to ambient air. Very little material indoors is strongly affected by SO_2. In contrast, estimates of cost associated with soiling by PM have emphasized residential household cleaning and maintenance, including a significant portion associated with indoor exposure. As noted by SRI (1981) and discussed in Chapter 5, indoor TSP levels do not correlate well with outdoor TSP levels. Clearly, then, to the extent that soiling is attributable to indoor PM levels, the relationship between soiling costs and outdoor PM levels (measured as TSP) is tenuous. A further difficulty in relating soiling to TSP is the fact that physical damage functions developed for the purpose have correlated loss of reflectance by the receptor surface to variation in TSP concentrations. When the receptors vary in texture and color as much as do the various building materials employed in developing damage functions for soiling, the resulting poor correlation (see Table 10-8) is inevitable.

The range of estimated economic loss associated with SO_x as displayed in Table 10-14 was discussed in Section 10.5.2.1. There is general agreement that damage for 1970 was approximately \$1 billion, plus or minus 50 percent. As noted earlier, there is evidence to suggest that paint damage costs should exclude the \$500 million developed from Spence and Haynie's association of TSP with exterior residential painting frequency. Similarly, exclusion of the costs associated with damage to dyes, to electrical switches and contacts, and to a number of materials included in the "other" category is indicated by physical evidence discussed previously. On the other hand, there is evidence of SO_2 effects on paint, on fabric strength, and building stone, concrete, and masonry. The Stankunas et al. (1981) study reported $31 million damages (1981 dollars) in SO_2-related paint erosion in the Boston area alone, although this is probably an overestimate, since it was based on a damage function developed for oil-based paint. Given no reliable basis for quantifying these effects, a reasonable course is to retain the $400 million "other" category in Table 10-14 to reflect damage to materials that has been shown to exist, but for which no national estimates of cost have been developed. Adopting this course, one can sum the remaining costs by category: $100 million for damage to industrial paints based on the Spence and Haynie (1972) study and $400 million for metal corrosion, totaling $500 million. Adding this to the $400 million for damage to fabrics, stone, concrete, and other paint results in a total estimate of $900 million for materials damage associated with SO_x levels in 1970. Assuming 50 percent error, the range is from $0.45 to 1.4 billion in 1978 dollars, with a midpoint of $0.9 billion.

Figure 10-11 illustrates how an estimate of benefits associated with improvements in SO_2 can be obtained. As shown in the figure, SO_2 levels have decreased by 44.5 percent during the period 1970 to 1978. If one assumes that the improvement is reflected linearly in decreased materials damage, the annual realized benefit for 1978 is approximately $0.4 billion.

Reported estimates of loss associated with soiling attributable to PM range up to $99 billion. Excluding the $99 billion estimated by Salmon (1970), who assumed that soiling of any material carried a cost, estimates of soiling damages in 1970 dollars generally are about $5 billion. Of the studies on household soiling attributable to airborne particles, three form a continuous development of a common data base.

As previously mentioned, the Booz, Allen and Hamilton study (1970) is the most comprehensive collection of data for cost and frequency of performing household cleaning and maintenance tasks. The inclusion of socioeconomic variables and reliable TSP data allowed for subsequent reanalysis by Watson and Jaksch (1978, 1982). Freeman's (1979b) estimate of $2.0 to 11.7 billion in 1978 dollars, equivalent to a range of $1.2 to 7.0 billion in 1970 dollars, although requiring adjustment to the 1982 Watson and Jaksch work, is the best synthesis of the data base developed by Booz, Allen and Hamilton (1970). In contrast, the Liu and Yu (1976) study was superficial in its treatment of the Booz, Allen and Hamilton data, and the Waddell estimate was based on the relationship of property value differentials to sulfation levels rather than TSP.

Freeman's range included benefits attributed to decreased indoor soiling and maintenance tasks. As SRI noted, and as is discussed in Chapter 5, the relationship between indoor particulate pollution and outdoor TSP levels is tenuous. Levels of outdoor particulate matter less than 2-3 μm in size, however, tend to be more strongly associated with levels of fine-mode indoor particulate matter, although specific information on this subject is limited.

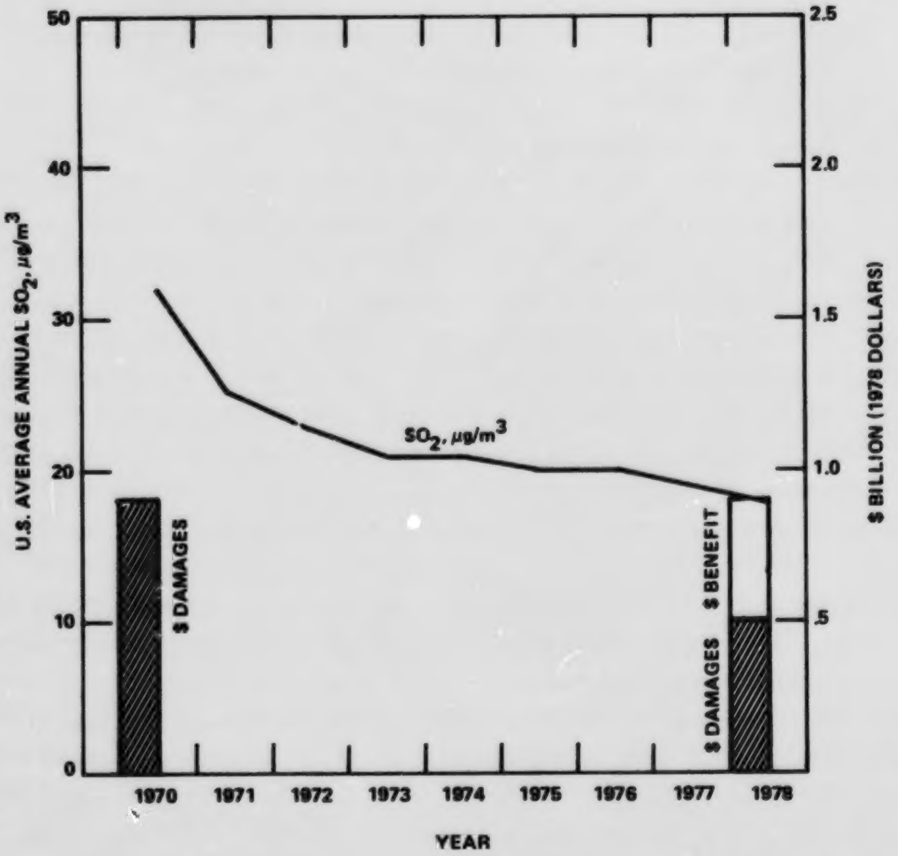


Figure 10-11. Improvement in U.S. annual average SO_2 levels from 32 $\mu g/m^3$ in 1970 to 18 $\mu g/m^3$ in 1978 has resulted in approximately $0.4 billion estimated economic benefit for 1978. Area labeled '$ DAMAGES' refers to 1978 dollar value of damage to materials attributable to sulfur oxides.

To arrive at some lower bound estimate of benefit associated with reduced soiling associated with decreased TSP levels, it seems prudent to confine such an estimate to outdoor residential soiling and maintenance. From Watson and Jaksch (1978, 1982), the observed cost outlays for cleaning and maintenance costs associated with outdoor pollution was 38 percent of the total outlays for both indoor and outdoor tasks. Applying the 38 percent factor to omit indoor pollution from the Watson and Jaksch estimates reduces the amount of total benefits from the range of \$2.4 to \$9.1 billion (Watson and Jaksch, 1982) to the range of \$1.0 to \$3.5 billion, or a best estimate of \$2.0 billion in 1978 dollars for achieving a level of 60 $\mu g/m^3$ TSP. Considering an air quality improvement of 20 percent for annual average TSP (72 $\mu g/m^3$ in 1970 to 58 $\mu g/m^3$ in 1978), the range of benefits for improved TSP levels during the period would be \$0.2 to \$0.7 billion, depending on which assumption of cleaning frequency is used.

One may, from the estimates of benefits resulting from improvement in TSP and SO_2 over the period 1970-1978, derive a rough notion of benefit per unit improvement in pollutant level by dividing the 14 $\mu g/m^3$ improvement in TSP or SO_2 by the associated annual benefit. This exercise results in an, approximate figure of \$30 million annual benefit for each $\mu g/m^3$ reduction in SO_2; the range for TSP is \$14 to \$50 million. However, as noted at the outset of this section, figures derived from a data base so replete with uncertainty can only be indicative of the general direction and magnitude of the change in benefits associated with changes in pollutant levels and by no means should be accepted as a reliable benefit estimate.

10.5.4 Summary of Economic Damage of Particulate Matter/Sulfur Oxides to Materials

The damaging and soiling of materials by airborne pollutants have an economic impact, but this impact is difficult to measure. The accuracy of economic damage functions is limited by several factors. One of the problems has been to separate costs related to SO_x and particles from each other and those related to other pollutants, as well as from those related to normal maintenance. Cost studies typically involve broad assumptions about the kinds of materials that are exposed in a given area and then require complex statistical analysis to account for a selected number of variables. Attitudes regarding maintenance may vary culturally, further confounding the problem of quantifying economic impact.

Studies have used various approaches to determine pollutant-related costs for extra cleaning, early replacement, more frequent painting, and protective coating of susceptible materials, as well as other indicators of the adverse economic effects of pollutants. No study has produced completely satisfactory results, and estimates of cost vary widely. In 1978 dollars, the estimated economic loss for 1970 SO_x damage was approximately \$0.9 billion; for TSP exterior soiling of residential structures, \$2 billion. Damage functions indicate that reductions in pollutants will decrease physical and, therefore, economic damage. While the data base and methodology for attribution of costs to SO_x and PM are incomplete at this time, it is possible to estimate gains from improvements in air quality from 1970 to 1978. In 1978 dollars, annual gain from SO_x reduction is estimated to be \$0.4 billion. Based on existing studies, the direct monetary benefits from further reduction in SO_x levels are likely to prove minor. The annual benefit of decreased soiling owing to declining levels of PM between 1970 and 1978 is estimated to be \$0.2 to \$0.7 billion in 1978 dollars.

10.6 SUMMARY AND CONCLUSIONS, EFFECTS ON MATERIALS

The nature and extent of damage to materials by SO_x and PM have been investigated by field and laboratory studies. Both physical and economic damage functions have been developed for specific damage/effect parameters associated with exposure to these pollutants. To date, only a few of these functions are relatively reliable in determining damage, while none has been generally accepted for estimating costs.

The best documented and most significant damage from SO_x and PM is the acceleration of metal corrosion and the erosion of paint. Erosion of building materials and stone due to SO_x is also established, but the importance of SO_x relative to other agents is not clear. Although evidence of damage to fibers (cotton and nylon), paper, leather and electrical components has been reported, reliable damage estimates have not.

Relatively accurate physical damage functions have been calculated for the effects of SO_2 on the corrosion of galvanized steel. Determination of variables such as time of wetness and surface configuration affect the applicability of the functions. Similar, but less accurate, functions have also been developed for estimating erosion rates of oil-based paints from exposure to SO_2. The large-scale replacement of oil-based paint by much more SO_2-resistant latex paint, however, makes these estimates obsolete. The least reliable of the "significant" damage functions are those for soiling from PM. The poorly understood deposition rates and poorly characterized chemical and physical properties related to reflectance make general application of the functions difficult, if not impossible.

The limitations of these and other physical damage functions hinder accurate estimates of total material damage and soiling. Coupled with these limitations is the lack of material exposure estimates. These problems presently preclude complete and accurate estimates of the costs of damage based on a physical damage function approach. Studies based on this approach estimated materials damage in 1970 attributable to SO_x as ranging from \$0.6 to 1.2 billion, in 1970 dollars. Estimates of soiling costs due to PM were based principally on other approaches and ranged up to \$99 billion. Best estimates of economic loss in 1970 attributable to SO_2 and TSP, in 1978 dollars, are \$0.9 billion in materials damage and \$2 billion in soiling, respectively. Improvements in SO_2 and TSP have resulted in estimated annual benefits of \$0.4 for SO_2 and \$0.2 to \$0.7 billion for TSP. These estimates are crude, but can serve to represent the direction and magnitude of changes in benefits associated with improvement in air quality. Approaches to cost estimation with data requirements different from those necessary for the physical damage function approach have been attempted. Whether these approaches yield results adequate for decisionmaking purposes is not clear at present.

10.7 REFERENCES

Abe, H., Y. Ishii, and H. Kato. Evaluation of atmospheric factors by analyses of corrosion products and surface deposits on copper plates. Rail. Tech. Res. Inst. 12:170-174, 1971.

Anderson, J. W. Sulphur in Biology. Studies in Biology 101, University Park Press, Baltimore, MD, 1978. pp. 17-19.

Arnold, L., D.B. Honeyborne, and C.A. Price. Conservation of natural stone. Chem. Ind. (London) 8:345-347, 1976.

Banov, A. Paintings and Coatings. Structures Publishing Co., Farmington, MI, 1973.

Barton, K. Protection Against Atmospheric Corrosion: Theories and Methods. John Wiley & Sons, New York, NY, 1976.

Barton, K. The influence of dust on atmospheric corrosion of metals. Werkst. Korros. 8/9:547-549, 1958.

Beaver, H. Committee on Air Pollution: Report. HMSO, London, England, 1954.

Beloin, N.J. Fading of dyed fabrics by air pollution., Text. Chem. Color. 4:77-82, 1972.

Beloin, N.J., and F.H. Haynie. Soiling of building materials. J. Air Pollut. Control Assoc. 25:399-403, 1975.

Benedict, H. L., C. J. Miller, and R. G. Olson. Economic impact of air pollutants on plants in the United States, 1971.

Biefer, G.J. Atmospheric corrosion of steel in the Canadian Arctic. Mater. Perform. 20:16-19, 1981.

Bennett, L.H., J. Kruger, R.L. Parker, E. Passaglia, C. Reimann, A.W. Ruff, and H. Yakowitz. Economic Effects of Metallic Corrosion in the United States. Part I: A Report to the Congress by the National Bureau of Standards. NBS Special Publication 511-1, U. S. Department of Commerce, National Bureau of Standards, Washington, DC, May 1978.

Bird, C.E. Corrosion behavior of galvanized sheet in relation to variation in coating thickness. Mater. Prot. and Perform., 16, 14-16, 1977.

Booz, Allen and Hamilton, Inc. Study to Determine Residential Soiling Costs of Particulate Air Pollution. APTD-0715, U.S. Department of Health, Education and Welfare, National Air Pollution Control Administration, Raleigh, NC, October 1970.

Brauns, E., and U. Kalla. Corrosion rate of plain carbon steels in land, sea, and industrial atmospheres. Stahl Eisen, 85:406-12, 1965.

Bresle, A. The corrosion of steel and the dangerous chlorides. Met. Finish. 74:23-25, 30, 1976.

Brysson, R.S., B.J. Trask, J.B. Upham, and S.G. Booras. The effects of air pollution on exposed cotton fabrics. J. Air Pollut. Control Assoc. 17:294-298, 1967.

Campbell, G.G., G.G. Shurr, D.E. Slawikowski, and J.W. Spence. Assessing air pollution damage to coatings. J. Paint Technol. 46:59-71, 1974.

Carey, W. F. Atmospheric deposits in Britain--a study of dinginess. Int. J. Air. Pollut. 2:1-26, 1959.

Cavender, J. H., W. M. Cox, M. Georgevitch, N. A. Huey, G. A. Jutze, and C. E. Zimmer. Interstate Surveillance Project: Measurement of Air Pollution Using Static Monitors. APTD-0666, U.S. Environmental Protection Agency, Research Triangle Park, NC, May 1971.

Chandler, K.A., and M.B. Kilcullen. Survey of corrosion and atmospheric pollution in and around Sheffield.. Br. Corros. J. 3:80-84, 1968.

Cowling, J.E., and M.E. Roberts. Paints, varnishes, enamels, and lacquers. In: Deterioration of Materials-Causes and Preventive Techniques. G. A. Greathouse and C. J. Wessel, eds., Reinhold Publishing Corp., New York, NY, 1954. pp. 596-645.

Crocker, T.D. Urban Air Pollution Damage Functions: Theory and Measurement. California University, Riverside, CA, 1970.

Del Monte, M.C. Sabbioni, and O. Vittori. Airborne carbon particles and marble deterioration. Atmos. Environ. 15:645-652, 1981.

Duncan, J.R., and D.J. Spedding. The mode of initial reaction of SO_2 at a metal surface. Corros. Sci. 14:241-249, 1974.

Eckhardt, F.G.W. Microorganisms and weathering of a sandstone monument. In: Environmental Biogeochemistry and Geochemistry, vol 2: The Terrestrial Environment, Proceedings of the Third International Symposium, Ministry of Sciences and Arts of Niedersachsen, Deutsche Forschungsgemeinschaft, Wolfenbøtel, Germany, 1977. W. E. Krumbein, ed., Ann Arbor Science Publishers, Inc., Ann Arbor, MI, 1978. pp. 675-686.

Esmen, N.A. A direct measurement method for dustfall. J. Air Pollut. Control. Assoc. 23:34-36, 1973.

Evans, U.R. Mechanism of rusting under different conditions. Br. Corros. J. 7:10-14, 1972.

Ergang, R., and M.B. Rockel. Corrosion resistance of stainless steels in the atmosphere--evaluation of the results of weathering tests of up to 10 years duration. Werkst. Korros. 26:36, 1975.

Exley, L.M. A practical review of residual oil firing problems and solutions. Combustion 41:16-23, 1970.

Faith, W.L. Effects of atmospheric pollutants. In: Encyclopedia of Environmental Science and Engineering, J. R. Pfafflin and E. N. Ziegler, eds. Gordon and Breach Science Publishers, New York, NY, 1976. pp. 219-223.

Fink, F.W., F.H. Buttner, and W.K. Boyd. Technical-Economic Evaluation of Air-Pollution Corrosion Costs on Metals in the U.S. APTD-0654. U.S. Environmental Protection Agency, Research Triangle Park, NC, February 1971.

Fleetwood, M.J. Corrosion--resisting metal. 1. Non-ferrous alloys for marine applications. Corros. Prev. Control 22:6-12, 1975.

Fochtman, E.G., and G. Langer. Automobile paint damaged by airborne iron particles. J. Air Pollut. Control Assoc. 6:243-247, 1957.

Freeman, A.M., III. The Benefits of Environmental Improvement, Theory and Practice. Johns Hopkins University Press, Baltimore, MD, 1979a.

Freeman, A.M., III. The Benefits of Air and Water Pollution Control: A Review and Synthesis of Recent Estimates. Report prepared for the Council on Environmental Quality. Bowdoin College, Brunswick, ME, 1979b.

Gajendragadkar, S.K. Effects of sulphur dioxide on materials. Chem. Age India 28:673-677, 1977.

Gauri, K.L. Effect of acid rain on structures. In: Acid Rain, Proceedings of the National Convention Workshop Session, American Society of Civil Engineers, Boston, Massachusetts, April 2-6, 1979. American Society of Civil Engineers, Boston, MA, 1979. pp. 55-75, Reprint 3598.

Gibbons, E.V. The Corrosion Behaviour of the Major Architectural and Structural Metals in Canadian Atmospheres. Summary of Ten-Year Results of Group I. NRCC 11630, National Research Council of Canada, Ottawa, Canada, October 1970.

Gillette, D.G. SO_2 and material damage. J. Air Pollut. Control Assoc. 25:1238-1243, 1975.

Gillies, K.A., and D.J. Youston. Social effects of atmospheric emissions. Presented at the 69th Annual Meeting, Air Pollution Control Association, Portland, Oregon, June 27-July 1, 1976. Paper no. 76-19.4.

Glass, N. Environmental Effects of Increased Coal Utilization: Ecological Effects of Gaseous Emissions from Coal Combustion. EPA-600/7-78-108, U.S. Environmental Protection Agency, Corvallis, OR, June 1978.

Guttman, H. Effects of atmospheric factors on the corrosion of rolled zinc. In: Metal Corrosion in the Atmosphere, a Symposium Presented at the Seventieth Annual Meeting, American Society for Testing and Materials, Boston, Massachusetts, June 25-39, 1967. ASTM Special Technical Publication no. 435, American Society for Testing and Materials, Philadelphia, PA, 1968. pp. 223-239.

Guttman, H., and P.J. Sereda. Measurement of atmospheric factors affecting the corrosion of metals. In: Metal Corrosion in the Atmosphere: a Symposium Presented at the Seventieth Annual Meeting, American Society for Testing and Materials, Boston, Massachusetts, June 25-30, 1967. ASTM Special Technical Publication no. 435, American Society for Testing and Materials, Philadelphia, PA, 1968. pp. 326-359.

Haagenrud, S., and B. Ottar. Long range transport of air pollution and corrosion effects. In: Proceedings of the Seventh Scandinavian Corrosion Congress, Trondheim, Norway, 1975. pp. 102-115.

Hamilton, J. D. A synthesis and critical review of methods for estimating the economic damage of air pollution. In: Proceedings of the 72nd Annual Meeting of the Air Pollution Control Association, June 1979.

Hancock, R.P., N.A. Esmen, and C.P. Furber. Visual response to dustiness. J. Air Pollut. Control Assoc. 26:54-57, 1976.

Hansen, J. Ailing Treasures. Sci. 80 [Eighty] 1:58-61, 1980.

Harker, A.B., F.B. Mansfeld, D.R. Strauss, and D.D. Landis. Mechanism of SO_2 and H_2SO_4 Aerosol Zinc Corrosion. EPA-600/3-80-018, U.S. Environmental Protection Agency, Research Triangle Park, NC, January 1980.

Hay, T.K., and G.G. Schurr. Moisture diffusion phenomena in practical paint systems. J. Paint Technol. 43:63-72, 1971.

Haynie, F.H. The economics of clean air in perspective. Mater. Prot. Perf. 13:33-38, 1974.

Haynie, F.H. Air pollution effects on stress induced intergranular corrosion of 7005-T53 aluminum alloy. In: Stress Corrosion - New Approaches. ASTM Special Technical Publication 610, American Society for Testing and Materials, Philadelphia, PA, 1976. pp. 32-43.

Haynie, F.H. Theoretical air pollution and climate effects on materials confirmed by zinc corrosion data. In: Durability of Building Materials and Components. P. J. Sereda and G. G. Levitan, eds., ASTM Special Technical Publication 691, American Society for Testing and Materials, Philadelphia, PA, 1980. pp. 157-175.

Haynie, F. H., J. W. Spence, and J. B. Upham. Effects of Gaseous Pollutants on Materials--A Chamber Study. EPA-600/3-76-015, U.S. Environmental Protection Agency, Research Triangle Park, NC, February 1976.

Haynie, F.H., J.W. Spence, and J.B. Upham. Effects of air pollutants on weathering steel and galvanized steel: a chamber study. In: Atmospheric Factors Affecting the Corrosion of Engineering Metals. S. K. Coburn, ed., ASTM Special Publication 646, American Society for Testing and Materials, Philadelphia, PA, 1978. pp. 30-47.

Haynie, F.H., and J.B. Upham. Correlation between corrosion behavior of steel and atmospheric pollution data. In: Corrosion in Natural Environments. ASTM Special Technical Publication 558, American Society for Testing and Materials, Philadelphia, PA, 1974. pp. 33-51.

Haynie, F.H., and J.B. Upham. Effects of atmospheric pollutants on the corrosion behavior of steels. Mater. Prot. Perform. 10:18-21, 1971.

Haynie, F.H., and J.B. Upham. Effects of atmospheric sulfur dioxide on the corrosion of zinc. Mater. Prot. Perform. 9:35-40, 1970.

Hershaft, A. Air pollution damage functions. Environ. Sci. Technol. 10:992-995, 1976.

Holbrow, G.L. Atmospheric pollution: its measurement and some effects on paint. J. Oil Color Chem. Assoc. 45:701-718, 1962.

Hoshizawa, K., and K. Koyata. Formation and their prevention of acid smuts at oil-fired burner. Nenryo Kyokaishi 49:656-665, 1970.

ITT Electro-Physics Laboratories, Inc. A Survey and Economic Assessment of the Effect of Air Pollutants on Electrical Components, vol 1, Sections 1 through 9. APTD-0797, U.S. Environmental Protection Agency, Research Triangle Park, NC, August 1971.

Ireland, F.E. Pollution by oxides of sulfur. Chem. Eng. (London) (221): CE 261-CE262, 1968.

Johnson, J.B., P. Elliott, M.A. Winterbottom, and G.C. Wood. Short-term atmospheric corrosion of mild steel at two weather and pollution monitored sites. Corros. Sci. 17:691-700, 1977.

Karraker, D.G. The kinetics of the reaction between sulfurous acid and ferric iron. J. Phys. Chem. 67:871-874, 1963.

Kingery, R.E. The extent of the paper problem in large research collections and the comparative costs of available solutions. In: Summary of a Conference, Permanent/Durable, American Library Association and Virginia State Library, Washington, DC, September 16, 1960. Virginia State Library Publications no. 16, Virginia State Library, Richmond, VA, 1960. pp. 36-41.

Kottori, M.S. The corrosion rates of metals exposed to pulp-mill and smelter emissions. Presented at the 73rd Annual Meeting, Air Pollution Control Association, Montreal, Quebec, June 22-27, 1980. Paper no. 80-67.4.

Krumbein, W.E., and C. Lange. Decay of plaster, paintings and wall material of the interior of buildings via microbial activity. In: Environmental Biochemistry and Geochemistry, vol. 2: The Terrestrial Environment, Proceedings of the Third International Symposium, Minister of Sciences and Arts of Niedersachsen and Deutsche Forschungsgemeinschaft, Wolfbøttel, Germany, 1977. W. E. Krumbein, ed., Ann Arbor Science Publishers, Inc., Ann Arbor, MI, 1978. pp. 687-697.

Kucera, V. Effects of sulfur dioxide and acid precipitation on metals and anti-rust painted steel. Ambio 5:243-248, 1976.

Larrabee, C.P. Mechanisms by which ferrous metals corrode in the atmosphere. Corrosion (Houston) 15:526, 1959.

Larrabee, C.P., and S.K. Coburn. The atmospheric corrosion of steels as influenced by changes in chemical composition. In: First International Congress on Metallic Corrosion, London, April 10-15, 1961. Butterworth and Co., Ltd., London, England, 1962. pp. 276-285.

Litvin, A. Clear coatings for exposed architectural concrete. J. PCA Res. Dev. Lab. 10:49-57, 1968.

Liu, B., and E.S. Yu. Physical and Economic Damage Functions for Air Pollutants by Receptor. EPA 600/5-76-011, U. S. Environmental Protection Agency, Corvallis, OR, September 1976.

Luckat, S. Investigations concerning the protection against air pollutants of objects of natural stone. Staub Reinhalt. Luft 32: May 1972.

Luckat, S. Stone deterioration at the Cologne Cathedral and other monuments due to action of air pollutants. In: Proceedings of the 4th International Clean Air Congress, Tokyo, 1976. Japanese Union Air Pollution Prevention Association, Tokyo, Japan, 1977. pp. 128-30.

Mäler, K.G., and R.E. Wyzga. Economic Measurement of Environmental Damage: A Technical Handbook. Organization for Economic Co-operation, Paris, France, 1976.

Mansfeld, F.B. Regional Air Pollution Study. Effects of Airborne Sulfur Pollutants on Materials. EPA-600/4-80-007, U.S. Environmental Protection Agency, Research Triangle Park, NC, January 1980.

Mansfeld, F., and J.V. Kenkel. Electrochemical monitoring of atmospheric corrosion phenomena. Corros. Sci. 16:111-122, 1976.

Manuel, Ernest H., Jr., R.L. Horst, Jr., K.M. Brennan, W.N. Lanen, M.C. Duff, J.K. Tapiero. Benefits Analysis of Alternative Secondary National Ambient Air Quality Standards for Sulfur Dioxide and Total Suspended Particulates: vol. I-V. EPA Contract No. 68-02-3392, Mathtech, Inc., Princeton, NJ, July 1981.

Matsushima, I., Y. Ishizu, T. Ueno, M. Kanazashi, and K. Horikawa. Effect of structural and environmental factors on the practical use of low-alloy weathering steel. Boshoku Gijutsu 23:177-182, 1974.

McBee, W.C. and T.A. Sullivan. Development of Specialized Sulfur Concretes. R.I. 8346, U.S. Department of the Interior, Bureau of Mines, Washington, D.C. 1979.

McCarthy, E. F., A. R. Stankunas, and J. E. Yocom. Benefit model for pollution effects on material TRC Environmental Consultants, Inc. 1981.

Michelson, I., and B. Tourin. Comparative methods for studying costs of air pollution. Public Health Rep. 81(6):505-511, 1966.

Michelson, I., and B. Tourin. Report on Study of Validity of Extension of Economic Effects of Air Pollution Damage from Upper Ohio River Valley to Washington, DC Area. Environmental Health and Safety Research Associates, August 1967.

Michelson, I., and B. Tourin. The household costs of living in polluted air versus the costs of controlling air pollution in the twin Kansas Cities Metropolitan Area. Report to the U.S. Public Health Service Conference on Air Pollution Abatement in the twin Kansas Cities Metropolitan Area, 1968. Environmental Health and Safety Research Associates, New Rochelle, NY, 1968a.

Michelson, I., and B. Tourin. The household cost of air pollution in Connecticut. Report to the Connecticut State Department of Health. Environmental Health and Safety Research Associates, New Rochelle, NY, 1968b.

Moore, J.C., and J.R. O'Leary. Evaluation of Structural Steel Coatings in Relation to Industrial Atmospheric Conditions. West Virginia Department of Highways, Research Project 23, West Virginia Department of Highways, Charleston, WV, January 1975.

Narayanan, R.L., and B.W. Lancaster. Household maintenance costs and particulate air pollution. Clean Air (Melbourne) 7:10-13, 1973.

National Research Council, Subcommittee on Air Particles. Airborne Particles. University Park Press, Baltimore, MD, 1979.

Newton, R.G. Cathedral chemistry--conserving the stained glass. Chem. Br. 10:89-91, 1974.

Niesel, K. The weathering of building materials in atmospheres containing sulfur oxides-a literature discussion. Fortsch. Mineral. 57:68-124, 1979.

Nriagu, J.O. Deteriorative effects of sulfur pollution on materials. In: Sulfur in the Environment. Part II: Ecological Impacts. John Wiley & Sons, Inc., New York, NY, 1978. pp. 1-59.

Overseas Public Nuisance Study Mission. Present Status of Public Nuisance in Electric Power Industry of Japan. Central Power Council, Tokyo, Japan, September, 1965.

Oyama, M., A. Moriyama, H. Miyazato, and H. Sakugawa. Examples on the acid-smut fallout. Okinawa-ken Kogai Eisei Kenkyusho-Ho 8:32-33, 1974.

Parker A. The destructive effects of air pollution on materials. In: Proceedings of the 22nd Annual Conference, National Smoke Abatement Society, Bournemouth, England, September 28, 1955. National Smoke Abatement Society, Brighton, England, 1955. pp. 1-15.

Potter, P. Chimney corrosion and acid smut emission. N.Z. J. Dairy Sci. Technol. 6:11-14, 1971.

Prenderleith, H.J. The Preservation of Leather Bookbindings. The British Museum, London, England, 1957.

Ridker, Ronald G. Economic Costs of Air Pollution. Praeger, 1967.

Riederer, J. Pollution damage to works of art. In: New Concepts in Air Pollution Research. Experientia Suppl. 20:73-85, 1974.

Robbins, R.C. Inquiry into the Economic Effects of Air Pollution on Electrical Contacts. Stanford Research Institute, Menlo Park, CA, April 1970.

Rowden, E. Acid corrosion and smut emission by kiln exhaust gases. Trans. Br. Ceram. Soc. 68:227-229, 1968.

Russell, C. A. How environmental pollutants diminish contact reliability. Insul. Circuits 22:43-44, 1976.

Salmon, R.L. Systems Analysis of the Effects of Air Pollution on Materials. APTD-0943, U.S. Department of Health, Education and Welfare, National Air Pollution Control Administration, Raleigh, NC, January 1970.

Salvin, V.S. Effect of air pollutants on dyed fabrics. J. Air Pollut. Control Assoc. 13:416-422, 1963.

Salvin, V.S. Survey and economic assessment of the effects of air pollution on textile fibers and dyes. Final Report, Contract no. PH-22-68-2, U.S. Department of Health, Education and Welfare. National Air Pollution Control Administration. Raleigh, NC, June 1970.

Sanyal, B., and G.K. Singhania. Atmospheric corrosion of metals: Part I. J. Sci. Ind. Res. Sect. B. 15:248-255, 1956.

Satake, J., and T. Moroishi. Various factors affecting atmospheric corrosion of steels. In: Proceedings of the Fifth International Congress on Metal Corrosion, Tokyo, Japan, May 1972. National Association of Corrosion Engineers, Houston, TX., 1974. pp. 744-749.

Schwarz, H. On the effect on magnetite on atmospheric rust and on rust under a coat of paint. Werkst. Korros. 23:648-663, 1972.

Schwenk, W., and H. Ternes. Rate of corrosion of plain carbon and low-alloy structural steels. Stahl Eisen 88:318-321, 1968.

Sengupta, M., and A.A. de Gast. Environmental deterioration and evaluation for dimension stone. Can. Min. Metall. Bull. 65:54-58, 1972.

Sereda, P.L. Weather factors affecting corrosion of metals. In: Corrosion in Natural Environments. ASTM Special Technical Publication 558, American Society for Testing and Materials, Philadelphia, PA, 1974. pp. 7-22.

Simpson, J.W., and P.J. Horrobin, eds. The Weathering and Performance of Building Materials. Wiley-Interscience, New York, NY, 1970. p. 286.

Sleater, G.A. Stone Preservatives: Methods of Laboratory Testing and Preliminary Performance Criteria. NBS Technical Note 941, U.S. Department of Commerce, National Bureau of Standards, Washington, DC, 1977.

Spedding, D.J., P.P. Rowlands, and J.E. Taylor. Sorption of sulphur dioxide by indoor surfaces: leather. J. Appl. Chem. Biotechnol. 21:68-70, 1971.

Spence, J.W., F.H. Haynie, and J.B. Upham. Effects of gaseous pollutants on paints: A chamber study. J. Paint Technol. 47:57-63, 1975.

Spence, J.W., and F.H. Haynie. Design of a laboratory experiment to identify the effects of environmental pollutants on materials. In: Corrosion in Natural Environments. STP 558. American Society for Testing and Materials, Philadelphia, PA, 1974. pp. 279-291.

Spence, J.W., and F.H. Haynie. Paint Technology and Air Pollution: A Survey and Economic Assessment. AP-103, U.S. Environmental Protection Agency, Research Triangle Park, NC, February 1972.

SRI International. An Estimate of the Nonhealth Benefits of Meeting the Secondary National Ambient Air Quality Standards. Report prepared for the National Commission on Air Quality. Menlo Park, CA, 1981.

Stankunas, A.R., D.F. Unites, and E.F. McCarthy. Air pollution damage to manmade materials. Presented at the 74th Annual Meeting, Air Pollution Control Association, Philadelphia, Pennsylvania, June 21-26, 1981. Paper no. 81-65.3.

Stanners, J.F. Use of environmental data in atmospheric corrosion studies. Br. Corros. J. 5:117-121, 1970.

Sullivan, T.A. and W.C. McBee. Development and Testing of Superior Sulfur Concretes. RI 8/60, U.S. Department of the Interior, Bureau of Mines, Washington, D.C. 1976.

Sulphur Institute. Sulphur concrete. Sulphur Res. Dev. 2:2-14, 1979.

Svoboda, M., B. Knapek, and H. Kilcoua. The effect of pigments on the penetration of sulfur dioxide into coatings. J. Oil Colour Chem. Assoc. 56:172174, 1973.

Sydberger, T. Influence of Sulphur Pollution on the Atmospheric Corrosion of Steel. University of Gothenburg, Department of Inorganic Chemistry, Gothenburg, Sweden, 1976. pp. 3-8.

Sydberger, T., and R. Ericsson. Laboratory testing of the atmospheric corrosion of steel. Werkst. Korros. 28:154-158, 1976.

Sydberger, T., and N.G. Vannerberg. The influence of the relative humidity and corrosion products on the adsorption of sulfur dioxide on metal surfaces. Corros. Sci. 12:775-784, 1972.

Tomashov, N.D. The Science of Corrosion. The Macmillan Co., New York, NY, 1966.

Tombach, I. Measurement of local climatological and air pollution factors affecting stone decay. Presented at the Conservation of Historic Stone Buildings and Monuments Conference, National Materials Advisory Board, National Research Council, National Academy of Sciences, Washington, DC, February 2-4, 1981.

Upham, J.B. Atmospheric corrosion studies in two metropolitan areas. J. Air Pollut. Control Assoc. 17:398-402, 1967.

Upham, J.B., F.H. Haynie, and J.W. Spence. Fading of Selected Drapery Fabrics by Air Pollutants. U.S. Environmental Protection Agency, Chemistry and Physics Laboratory, Research Triangle Park, NC, 1975.

Upham, J.B., and V.S. Salvin. Effects of Air Pollutants on Textile Fibers and Dyes. EPA-650/3-74-008, U.S. Environmental Protection Agency, Research Triangle Park, NC, February 1975.

Verdu. Effect of air pollutants on aging of plastic materials. Trib. CEBEDEAU 27:360-370, 1974.

Vero, L.B., and M.M. Sila. Isolation of various sulphur-oxidizing bacteria from stone monuments. In: The Conservation of Stone, Proceedings of the International Symposium, Bologna, Italy, June 19-21, 1976.

Waddell, T.E. The Economic Damages of Air Pollution. EPA-600/5-74-012, U.S. Environmental Protection Agency, Washington, DC, May 1974.

Walsh, M., A. Black, A. Morgan, and G. Crashaw. Sorption of SO_2 by indoor surfaces including carpets, wallpaper, and paint. Atmos. Environ. 11:1107-1111, 1977.

Watson, W.D., Jr., and J.A. Jaksch. Household cleaning costs and air pollution. Presented at the 71st Annual Meeting, Air Pollution Control Association, Houston, Texas, June 25-30, 1978. Paper no. 78-52.3.

Watson, W.D., and J.A. Jaksch. Air pollution: household soiling and consumer welfare losses, 1982 (Paper to be published in the Journal of Environmental Economics and Management, Volume 9, No. 3, September 1982).

Winkler, E.M. Stone: Properties, Durability in Man's Environment. 2nd rev. ed., Springer-Verlag, New York, NY, 1975.

Winkler, E.M. Important agents of weathering for building and monumental stone. Eng. Geol. (Amsterdam) 1:381-400, 1966.

Yocom, J.E. and Grappone. Effects of Power Plant Emissions on Materials. EPRI EC-139, Electric Power Research Institute, Palo Alto, CA, July 1976.

Yocom, J.E. Air pollution damage to buildings on the Acropolis. J. Air Pollut. Control Assoc. 29:333-338, 1979.

Yocom, J.E., and R.O. McCaldin, Effects of air pollution on materials and the economy. In: Air Pollution, Vol. I, A. Stern, ed. Academic Press, New York, NY, 1968, 617-654.

Yocom, J.E., and J.B. Upham. Effects on economic materials and structures. In: Air Pollution. 3rd ed. vol. II: The Effects of Air Pollution. A. C. Stern, ed., Academic Press, Inc., New York, NY, 1977. pp. 65-116.

Yoshihara, K., T.C. Huang, H. Ebihora, and H. Shibata. Chemical effects of the $^{35}Cl(n,p)^{35}S$ reaction. Radiochim. Acta 3:185-191, 1964.

Zeronian, S.H. Reactions of cellulosic fabrics to air contaminated with sulfur dioxide. Text. Res. J. 40:695-698, 1970.

Zeronian, S.H., K.W. Alger, and S.T. Omaye. Reactions of fabrics made from synthetic fibers to air contaminated with nitrogen dioxide, ozone, or sulfur dioxide. In: Proceedings of the Second International Clean Air Congress, International Union of Air Pollution, Washington, DC, December 6-11, 1970. H. M. Englund and W. T. Beery, eds., Academic Press, New York, NY, 1971. pp. 468-476.

11. RESPIRATORY TRACT DEPOSITION AND FATE OF INHALED AEROSOLS AND SULFUR DIOXIDE

11.1 INTRODUCTION

11.1.1 General Considerations

The respiratory system is the major route of human exposure to airborne suspensions of particles (aerosols) and sulfur dioxide (SO_2) gas. During inhalation (and exhalation), a portion of the inhaled aerosol and gas may be deposited by contact with airway surfaces, or it may be transferred to unexhaled air; the remainder is exhaled. The portion transferred to unexhaled air may either be deposited by contact with airway surfaces or exhaled later. These phenomena are complicated by interactions that may occur among the particles, the SO_2 gas, other gases such as endogenous ammonia, and the water vapor present in the airways.

In inhalation toxicology, specific terminology is applied to these processes. The term deposition refers specifically to the removal of inhaled particles or gases by the respiratory tract and to the initial regional pattern of these deposited materials. The term clearance refers to the subsequent translocation (movement of material within the lung or to other organs), transformation, and removal of deposited substances from the respiratory tract or from the body. It can also refer to the removal of reaction products formed from SO_2 or particles. The temporal pattern of uncleared deposited particulate materials or gases and reaction products is called retention.

The mechanisms involved in the deposition of inhaled aerosols and gases are affected by physical and chemical properties, including aerosol particle size distribution, density, shape, surface area, electrostatic charge, hygroscopicity or deliquescence, chemical composition, gas diffusivity and solubility, and related reactions. The geometry of the respiratory airways from nose and mouth to the lung parenchyma also influences aerosol deposition; the important morphological parameters include the diameters, lengths, inclinations to vertical, and branching angles of airway segments. Physiological factors that affect deposition include breathing patterns, airflow dynamics in the respiratory tract, and variations of relative humidity and temperature within the airways. Clearance from the respiratory tract depends on many factors, including site of deposition, chemical composition and properties of the deposited particles, reaction products, mucociliary transport in the tracheobronchial tree, macrophage phagocytosis, and pulmonary lymph and blood flow. An understanding of the regional deposition and clearance of particles and SO_2 is essential to the interpretation of the results of health effects studies described in Chapters 12-14.

Translocation of sulfur compounds or other materials from the lung to other organs is also important, since the lung can be the portal of entry for toxic agents that affect other organs of the body. Hence, multicompartment models of clearance from the respiratory tract to other organs can provide predictive information about the potential for injury of those other organs. Mathematical representations of lung retention and translocation require data on the various factors that affect deposition and clearance.

Since many conclusions concerning the deposition, clearance, and health impact of inhaled aerosols and SO_2 are based on data obtained from animal experiments, care must be taken to identify physiological and anatomical differences between human beings and animals which may influence these phenomena. The following discussion will emphasize the regional deposition and clearance that occur in human airways, but selected comparisons will be made with other mammalian species to clarify differences that may affect health-impact analyses of experimental data.

11.1.2 Aerosol and Sulfur Dioxide Characteristics

An aerosol may be defined as a relatively stable suspension of liquid or solid particles in a gaseous medium (see Chapter 2 for a detailed discussion of the physicochemical properties of aerosols and SO_2). Airborne particulate materials in the environment are aerosols with a variety of physical and chemical properties. In particular, a given aerosol may include particles with a wide spectrum of physical sizes, even if all the particles have similar chemical composition; the concentration of toxic components in particles may be different for different sized particles (Natusch et al., 1974); or morphologically identical particles may have totally different chemical compositions (Pawley and Fisher, 1977). Common assumptions that particles in a given aerosol have a relatively homogeneous chemical composition, toxic potential, and physical density may be seriously misleading, especially when particles are found in combination with SO_2 gas.

The relevant physical and chemical properties of aerosols and gases must be characterized appropriately to evaluate the effect of their inhalation on health. These properties then can provide predictive information concerning regional respiratory tract deposition and other important dosimetric factors that need to be considered if biological responses described in Chapters 12-14 are to be understood adequately.

If particles in an aerosol are smooth and spherical or nearly spherical, their physical sizes can be conveniently described in terms of their respective geometric diameters. Even unagglomerated aerosols of solids, however, rarely contain smooth, spherical particles. Various conventions for describing physical diameters have been based on available methods of observing and measuring particle size. For example, the size of a particle may be described in terms of its projected area diameter (D_p), defined as the diameter of a circle with an area equal to the apparent cross-sectional area of the particle when lying on a collection surface and viewed with an optical or electron microscope. Other conventions for describing physical size are based on measurements of scattered light, surface area, electrical mobility, diffusional mobility, or other physical or chemical phenomena (Mercer, 1973; Stockham and Fochtman, 1979).

The aerodynamic properties of aerosol particles depend on a variety of physical parameters including size, shape, and physical density. Two important aerodynamic properties of aerosol particles are the inertial properties, which are most important for particles larger than 0.5 μm in diameter and are related to the settling speed in air under the influence of the earth's gravity, and the diffusional properties, which are most important for particles smaller than 0.5 μm in diameter and are related to the diffusion coefficient (Fuchs, 1964) (see Section 11.2.1). When particles are inhaled, their aerodynamic properties, combined with various anatomical and breathing characteristics, determine their fractional deposition in various regions in the respiratory tract.

To avoid the complications associated with the effects of particle shape, size, and physical density on the inertial properties of inhaled airborne particles, "aerodynamic diameters" have been defined and used to describe particles with common inertial properties with the same "aerodynamic diameter." The aerodynamic diameter most generally used is the aerodynamic equivalent diameter (D_{ae}), defined by Hatch and Gross (1964) as "the diameter of a unit density sphere having the same settling speed (under gravity) as the particle in question of whatever shape and density." Raabe (1976) recommended the use of an aerodynamic resistance diameter (D_{ar}), defined more directly with terms used in physics to describe the inertial properties of a particle. The difference between these two diameters is only 0.08 μm or less over all sizes under normal conditions at sea level. Hence, the term aerodynamic diameter can be used to refer to either or both of these two definitions.

Environmental aerosols have size distributions that are more complicated, reflecting the production of particles by atmospheric processes, emission sources or other anthropogenic activities, and the particle dynamics. They may have several modes (Whitby, 1978). Photochemical reactions and certain combustion processes create small particles that are generally smaller than 0.1 μm (the nuclei mode), whereas other combustion, condensation, and mechanical particle generation processes yield larger particles. Another mode, between 0.1 and 2 μm, is known as the accummulation mode and includes primary emissions plus aggregates and droplets formed by coagulation of the primary nuclei mode particles and the materials that condense on them from the vapor phase. Coarse particles, larger than about 2 um, are formed primarily by mechanical processes. The particle size distribution within each of the three modes (nuclei, accumulation, and coarse) is generally lognormal, as defined below.

Since not all particles in an aerosol are of the same physical or aerodynamic size, the distribution of sizes must be described. If either the physical diameter (D) or the aerodynamic diameter is used to characterize particles, the distribution of particle sizes in a mixed aerosol is most conveniently described as a probability density function. One such generally useful function, the lognormal function, involves the geometric mean size (or median) and the geometric standard deviation (σ_g) and refers to a normal distribution with respect to the logarithm of particle diameter. Hence, if the particle number is being considered, the particle size may be reported as the count median (physical) diameter (CMD) and σ_g. Half of the number of particles in an aerosol has physical sizes less than the CMD and half has larger. Since the mass of a material is usually more relevant to its potential toxicity, the mass median (geometrical) diameter (MMD) or mass median aerodynamic diameter (MMAD) and σ_g is usually preferred in describing aerosols in inhalation toxicology research. Half the mass of particles in an aerosol is associated with particles smaller than the MMD and half with larger particles. Likewise, half the mass of particles is associated with particles whose aerodynamic diameters are smaller than the MMAD and half with particles with larger aerodynamic diameters. If an aerosol is radioactive or radiolabeled, mass measurements may be replaced by activity measurements. Interrelationships among these various ways to express the diameter of the aerosol were examined for the lognormal distribution by Raabe (1971).

In addition to particle characteristics, conditions of the gas medium influence the properties of aerosol dispersions. Such environmental conditions as relative humidity, temperature, barometric pressure, and fluid flow conditions (e.g., wind velocity or state of turbulence) affect the aerodynamics of aerosol particles.

The concentration of environmental aerosols or gases generally does not affect inhalation deposition and particle dynamics. The mass concentration (mg/m^3 or $\mu g/m^3$) or concentration of a specific potentially toxic species (mg of constituent/m^3) provides information needed to calculate inhalation exposure levels. For SO_2, the concentration may be expressed in parts per million (ppm) by volume or in mass concentrations (mg/m^3); each 1 ppm of SO_2 equals 2.62 mg/m^3 (2620 $\mu g/m^3$) at an air temperature of 25°C.

Sulfur dioxide gas is a rapidly diffusing reactive gas that is readily soluble in water and body fluids (Aharonson, 1976). This property is responsible for the extensive removal of SO_2 in the extrathoracic region and in the upper generations of the tracheobronchial tree. Extraction of SO_2 during nose breathing is significantly greater than during mouth breathing, and over a 4- to 6-hour exposure to high levels of SO_2, no saturation effect for absorption can be seen (see Section 11.2.4). Through normal and catalyst-mediated oxidation processes in air, SO_2 gas is slowly oxidized to sulfite (SO_3) that rapidly hydrolyzes to form sulfuric acid (H_2SO_4), leading to sulfate salts. Since ammonia (NH_3) is formed in natural biological processes including endogenously in the airways, $(NH_4)_2SO_4$ and NH_4HSO_4 are important products of H_2SO_4 neutralization.

11.1.3 The Respiratory Tract

The respiratory tract (Figure 11-1) includes the passages of the nose, mouth, nasal pharynx, oral pharynx, epiglottis, larynx, trachea, bronchi, bronchioles, and small ducts and alveoli of the pulmonary acini. With respect to respiratory tract deposition and clearance of inhaled aerosols, three regions can be considered: (1) extrathoracic (ET), the airways extending from the nares down to the epiglottis and larynx at the entrance to the trachea (the mouth is included in this region during mouth breathing); (2) tracheobronchial (TB), the primary conducting airways of the lung from the trachea to the terminal bronchioles (i.e., that portion of the lung respiratory tract having a ciliated epithelium); and (3) pulmonary (P), the parenchymal airspaces of the lung, including the respiratory bronchioles, alveolar ducts, alveolar sacs, atria, and alveoli (i.e., the gas-exchange region). The extrathoracic region, as defined above, corresponds exactly to the nasopharynx, as defined by the International Commission on Radiological Protection (ICRP) Task Group on Lung Dynamics (Morrow et al., 1966).

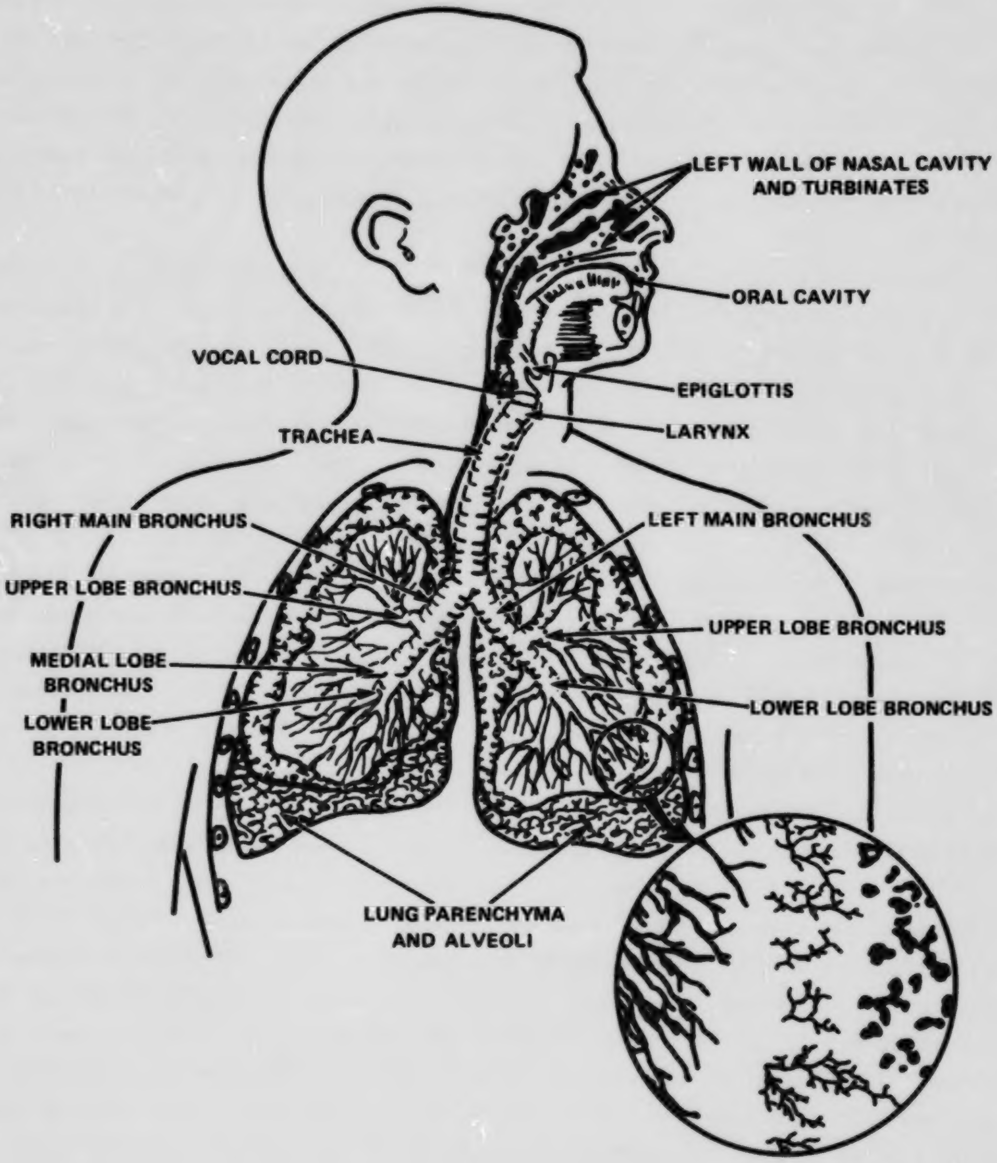


Figure 11-1. Features of the respiratory tract of man used in the description of the deposition of inhaled particles and gases with insert showing parts of a silicon rubber cast of a human lung showing some separated bronchioles to 3 mm diameter, some bronchioles from 3 mm diameter to terminal bronchioles, and some separated respiratory acinus bundles.

Source: Adapted from Hatch and Gross (1964) and Raabe (1979).

The nose is a complex structure of cartilage and muscle supported by bone and lined with mucosa (Holmes et al., 1950). The vestibule of the nares is unciliated but contains a low-resistance filter consisting of small hairs. The nasal volume is separated into two cavities by a 2- to 7-mm thick septum. The inner nasal fossae and turbinates are ciliated, with mucus flow in the direction of the pharynx. The turbinates are shelf-like projections of bone covered by ciliated mucous membranes with a high surface-to-volume ratio that facilitate humidification of the incoming air. The larynx consists of two pairs of mucosal folds that narrow the airway.

The trachea, an elastic tube supported by 16 to 20 cartilagenous rings that circle about three-fourths of its circumference, is the first and largest of a series of branching airway ducts (Tenney and Bartlett, 1967). The left and right lungs are entered by the two major bronchi that branch off of the trachea (see Figure 11-1). The left lung consists of two clearly separated upper and lower lobes; the right lung consists of the upper, middle, and lower lobes. The conductive airways in each lobe of the lung consist of up to 18 to 20 dichotomous branches from the bronchi to the terminal bronchiole (Pump, 1964; Raabe et al., 1976).

The pulmonary gas-exchange region of the lung begins with the partially alveolated respiratory bronchioles. Pulmonary branching proceeds through a few levels of respiratory bronchioles to completely alveolated ducts (Smith and Boyden, 1949; Whimster, 1970; Krahl, 1963) and alveolar sacs (Tenney and Remmers, 1963; Pattle, 1961b; Machlin, 1950; Fraser and Pare, 1971). Alveoli are thin-walled polyhedral air pouches that cluster about the acinus through connections with respiratory bronchioles, alveolar ducts, or alveolar sacs.

The airway spaces in the pulmonary region are coated with a complex aqueous liquid containing several biochemically specialized substances (Green, 1974; Blank et al., 1969; Balis et al., 1971; Pattle, 1961b; Kott et al., 1974; Henderson et al., 1975; Kanapilly, 1977). An understanding of the chemical composition and dynamic nature of the acellular layer at the air-alveolar surface is needed to comprehend the general behavior of material deposited in the pulmonary region. This acellular layer consists of a surfactant film < 0.01 µm thick and a hypophase about 0.1 to 0.2 µm thick (Clements and Tierney, 1965). The thickness of the complex aqueous lining layer is not uniform because pools of surfactant even out pits, crevices, folds and small surface irregularities, and thereby help to impart smoothness to the alveolar surface (Gil et al., 1979). A mixture of phospholipids and neutral lipids is contained in the surfactant film (Scarpelli, 1968; Pfleger and Thomas, 1971; Pruitt et al., 1971; Reifenath, 1973). The major phospholipid is dipalmitoyl lecithin, and the major neutral lipids are cholesterol and its esters. Protein content in lung surfactant is less than 20 percent by weight (Pruitt et al., 1971; Klass, 1973). The composition of the hypophase is not well understood, with lung surfactant materials, mucopolysaccharides, lipoproteins, and possibly serum proteins such as albumin likely to be present (Scarpelli, 1968; Reifenath, 1973; Tuttle and Westerberg, 1974). The pH of alveolar fluid may be similar to that of blood fluid. Various factors favoring and opposing transudation of fluid across the air-blood barrier result in the cyclic movement of fluid in and out of alveoli, thereby helping to maintain the very thin layer of alveolar fluid (Kanapilly, 1977). The concentrations of chelating and precipitating agents in the alveolar fluid influence the retention and transport of particles deposited in the pulmonary region; however, the concentrations of chelating agents are not sufficiently high to prevent the transformation of polyvalent cations into an insoluble form (Kanapilly, 1977).

The parenchyma of the pulmonary region includes several types of tissue, circulating blood, lymphatic drainage pathways, and lymph nodes. In humans, the weight of the lung, including circulating blood, is about 1.4 percent of the total body weight. The weight of lung blood is equal to about 0.7 percent of total body weight (10 percent of total blood volume) (International Commission of Radiological Protection, 1975). Because a large portion of lung is occupied by air, the average physical density of the parenchyma is about 0.26 g/cm^3 (Fowler and Young, 1959).

Models of the airways, which simplify the complex array of branching and dimensions into workable mathematical functions, are useful in comparing theoretical predictions of deposition with experimentally-obtained deposition data, thereby leading to more refined models and increasing our understanding of the processes that affect respiratory tract deposition. An early idealized model of the airways of the human lung was developed by Findeisen (1935) for estimating the deposition of inhaled particles. Findeisen's model assumed branching symmetry within the lung, with each generation consisting of airways of identical size. Landahl (1950), Davies (1961), Weibel (1963), and Horsfield and Cumming (1968) proposed other models based on a symmetry assumption. Asymmetric models that more closely approximate the human lung were developed by Weibel (1963), Horsfield et al. (1971), and Horsfield and Cumming (1968). Yeh and Schum (1980) proposed a typical pathway lung model and made particle deposition calculations for each lobe of the lungs. Although particle deposition models currently available ignore the dynamic nature of the airways, future models should consider this aspect.

11.1.4 Respiration and Other Factors

Both the humidity and temperature of inhaled aerosols and gases, as well as the subsequent changes that occur as the aerosol-gas mixture passes through various parts of the airways, influence the inhalation deposition of airborne particles. Deposition of hygroscopic aerosols will depend in part on the relative humidity in the airways, since the growth of such particles will directly affect both the site and extent of inhalation deposition (see Section 11.2.2).

The complex anatomical structure of the nose is well suited for humidification, regulation of temperature, and removal of many particles and gases. The relative humidity of inhaled air probably reaches near saturation in the nose (Verzar et al., 1953). Since the human nose is a short passageway, tranquil diffusion alone cannot account for rapid humidification. Rather, convective mixing must play a role, suggesting a mechanism for enhancing SO_2 collection in the nose. The temperature of the inhaled air may not reach body temperature until relatively deep in the lung. Deal et al. (1979a,b,c) measured retrocardiac and retrotracheal temperatures under different ambient temperatures and found airway cooling associated with breathing cool air. Raabe et al. (1976) found that the temperature of the air at major bronchi in a nose-breathing dog averaged 35°C, 4°C less than the body temperature.

The air-deflecting channels of the anterior nares cause impaction of large airborne particles and create turbulent airflow conditions. As the cross-sectional area expands beyond the entrance, flow separation occurs resulting in turbulence and eddies, which continue as the air traverses the passages around the turbinates. Proctor and Swift (1971) studied the flow of water through a clear plastic model of the walls of the nasal passages and constructed charts of the direction and linear velocity of airflow from this model. With a steady inspiratory flow of 0.4 l/sec, the investigators found that the linear inspiratory velocity at the nasal entrance reached at least 4.5 to 5 m/s and at most 10 to 12 m/s, values that are significantly greater than the 2 m/s peak linear velocity in the tracheobronchial tree during quiet breathing.

The caliber of the trachea and major bronchi and their cross-sectional geometry is about 15 percent larger during inspiration than during expiration (Marshall and Holden, 1963; Fraser and Pare, 1971; Raabe et al., 1976), although the caliber of the smaller conductive bronchioles may be up to 40 percent greater during inspiration (Marshall and Holden, 1963; Hughes et al., 1972). Bronchial caliber correlates with body size (Thurlbeck and Haines, 1975).

Schroter and Sudlow (1969) studied a wide variety of flow patterns and flowrates in large-scale symmetrical models of typical tracheobronchial tree junctions. For both inspiration and expiration and irrespective of entry profile form, they observed secondary flows at all flowrates in their single bifurcation model. When a second bifurcation was added a short distance downstream of the first, the entering flow profile was found to influence the resulting flow patterns. Also, different results were obtained depending on the plane in which the second bifurcation was located relative to the first bifurcation.

Olson et al. (1973) studied convective airflow patterns in cast replicas of the human respiratory tract during steady inspiration. They showed that the effect of the larynx is such that flow patterns typical of smooth bifurcating tubes do not occur until the lobar bronchi are reached. Small eddies were observed as far down as the sublobar bronchi with flows of 200 ml/s in the trachea. In humans, the glottis of the larynx acts as a variable orifice, since the position of the vocal cords changes. During inspiration, a jet of turbulent air enters the trachea and is directed against its ventral wall, imparting additional turbulence over that associated with the corrugated walls and length of the trachea.

In the tracheobronchial tree, with its many branches, changes in caliber, and irregular wall surfaces, establishing exactly where flow is laminar, turbulent, or transitional is difficult. Viscous forces predominate in laminar flow, and streamlines persist for great distances; with turbulent flow, there is rapid and random mixing downstream. As the flowrate increases, unsteadiness develops and separation of the streamlines from the wall can occur, leading to the formation of local eddies. This type of flow is termed transitional. The Reynolds number, the ratio of inertial to viscous forces, is useful in describing whether flow is laminar or turbulent. In smooth-walled tubes, values between approximately 2000 and 4000 are ascribed to transitional flow, with smaller Reynolds numbers reflecting laminar flow and larger ones turbulent flow. Fully developed laminar flow probably only occurs in the very small airways; flow is transitional in most of the tracheobronchial tree, although true turbulence may occur in the trachea, especially during exercise when flow velocities are high (West, 1977).

Turbulence will gradually decay in any branch in which the Reynolds number is less than 3000 (Owen, 1969). Decays of 15, 16, and 10 percent are predicted to occur in the first three generations of bifurcation, respectively, using the theory of Batchelor (1953) for the change in turbulent energy at regions of rapid flow contraction. Although these decay calculations neglect the possible effects of the strong secondary flows generated at the bifurcation, their validity is supported by the data of Pedley et al.(1971) which show that the boundary layer remains laminar in the daughter-tube for Reynolds numbers in the parent-tube up to at least 10,000. Hence, the turbulent eddies are localized in the center.

Flow oscillations in the segmental bronchi attributed to the beating of the heart are only detectable during breathholding or during pauses between inspiration and expiration (West, 1961). A peak oscillatory flowrate of 0.5 l/min was measured, which is about 20 percent of the peak flowrate in the segmental bronchi during quiet breathing. Gas mixing is improved by these oscillations.

Gas flow dynamics may be expected to be turbulent within the upper airways of humans and dogs but laminar everywhere in the airways of small rodents (Dekker, 1961; Fry, 1968; Schroter and Sudlow, 1969; Olson et al., 1973; Martin and Jacobi, 1972; West, 1961). The larynx introduces an important airflow disturbance that can influence tracheal deposition (Bartlett et al., 1973; Schlesinger and Lippmann, 1976). In the smaller human bronchi and bronchioles where fluid flow is relatively tranquil, laminar flow prevails; but branching patterns, filling patterns (Grant et al., 1974), flow reversals with varying velocity profiles, and swirling complicate a description of flow in the small airways (Silverman and Billings, 1961; Cinkotai, 1974). Because actual flow in the respiratory airways is difficult to describe, simplifying assumptions, such as parabolic laminar or uniform velocity profiles, are usually incorporated into analytic descriptions.

Inspiratory flowrate and depth of inhalation influence the deposition of inhaled particles. The air inspired in one breath is the tidal volume (TV). The average inspiratory flowrate (Q) and TV (Bake et al., 1974; Clement et al., 1973) affect both inertial and diffusional deposition processes (Altshuler et al., 1967). The total air remaining in the lungs at the end of normal expiration affects the relative mixing of inhaled particles and, when compared with total lung capacity, is indicative of the extent of aerosol penetration into the lung (West, 1974; Luft, 1958). Guyton (1947a,b) and Stahl (1967) developed interspecies relationships describing respiratory volumes and patterns.

The inspiratory capacity, the maximum volume of air that can be inhaled after a given normal expiration, is contrasted to the vital capacity, which is the maximum volume of air that can be expelled from the lungs with effort after maximum forced inspiration. Air that remains in the conductive airways (from nose or mouth to terminal bronchioles) at the end of expiration is considered to occupy the anatomical dead space, since the conductive airways are not involved in gas exchange.

Representative values for normal human respiratory parameters, which can be used for deposition and dosimetric predictions, are available from various sources (Zenz, 1975; Higgs et al., 1967; American Heart Association, 1973; Jones et al., 1975; Intermountain Thoracic Society, 1975; International Commission on Radiological Protection, 1975). Considerable variability in respiratory parameters may occur among individuals in the population, particularly when healthy adults are contrasted with children, the aged, and ill individuals. Average TV has a reasonably fixed relationship with body weight of 7 to 10 ml/kg from birth to adulthood (Doershuk et al., 1970, 1975). The gas-exchange area increases proportionally with age and more or less with height, but not with body surface area. Average values for the gas-exchange area are 6.5, 32, and 75 m^2 at 3 mo, 8 y, and adulthood, respectively (Dunnill, 1962). Respiratory frequency decreases from about 35 breaths per minute (BPM) at birth to 12 to 16 BPM with normal respiration in adulthood (Polgar and Weng, 1979).

In some instances, the total and regional deposition data presented in Section 11.2 exhibit considerable scatter. Some of this variability might be expected given the range of breathing frequencies, TV's, and average inspiratory flowrates used in the various deposition experiments. Deposition studies involving aerosol persistence during breath-holding led Lapp and coworkers (1975) to conclude that marked differences exist in airway geometry among subjects with similar heights and lung volumes. An interlaboratory comparison study of lung deposition data by Heyder et al. (1978), besides identifying possible sources of errors connected with the experimental technique, identified different deposition data in the subjects. Further data on intersubject variability, including data on variability of regional deposition, were presented by Stahlhofen et al. (1981). Yu and coworkers (1979) used Monte Carlo techniques to determine the total and regional deposition of inhaled particles in a population of human lungs by taking into account variability in airway dimensions. Their results for particle sizes ranging from 0.1 to 8 µm D_{ae} suggest that observed subject deposition variability is caused primarily by differences in airway dimensions. When the total respiratory tract deposition of particles between 0.3 and 1.5 µm D_{ae} is studied, expressing the data as a function of the ratio of the relative expiratory reserve volume to the normal expiratory reserve volume greatly reduces intersubject variability (Tarroni et al., 1980).

The vast majority of studies on the deposition of particles in humans has been conducted in young healthy adults. Consequently, there is a paucity of data on deposition (and clearance) in other subpopulations, such as children, asthmatics, chronic bronchitics, etc. Significant pathologic changes in airways and parenchyma can markedly alter the deposition of particles. For example, Lippmann et al. (1971) found substantially increased bronchial deposition in chronic bronchitic and asthmatic subjects. These increases may vary with different phases of the disease (Goldberg and Lourenco, 1973). Tracheobronchial deposition appears to be enhanced at the expense of pulmonary deposition in most abnormal states. For example, the deposition of 2 μm particles in patients with bronchiectasis is frequently more central than that in normal subjects (Lourenco et al., 1972). Partial or complete airway obstruction in bronchitis, lung cancer, emphysema, fibrosis, and atelectasis may decrease or eliminate deposition of particles in some regions of the lungs (Taplin et al., 1970). The numerous and complex mechanisms responsible for alterations in the pattern of deposition in various disease states need to be studied.

Currently available human deposition data have been collected from volunteers inhaling aerosols through either mouthpieces or nose masks. Differences in mass burden of particles and gases between these controlled inhalations and normal, spontaneous mouth breathing or nose breathing are possible; however, Heyder et al. (1980a) found the same dependence on a deposition parameter incorporating minute volume, breathing frequency, and particle size for both spontaneous and artificially controlled breathing patterns in three subjects. Any differences are more likely to be seen at minute ventilations corresponding to heavy or maximal exercise, since oral resistance to respiratory airflow is probably lowered when a subject breathes through a mouthpiece. Studies in which the nose of the subject is completely occluded with a clip do not simulate oronasal breathing because no air passes through the nose and the oral airway is wider than usual. With partial nasal obstruction or in exercise, most human beings resort to oronasal breathing. In studies designed to examine the switching point from nasal to oronasal breathing (Niinimaa et al., 1980) and to determine the oronasal distribution of respiratory airflow in 30 healthy adult subjects (14 males, 16 females) (Niinimaa et al., 1981), 20 of the subjects (67 percent) switched from nasal to orally augmented breathing with exercise and 4 subjects (13 percent) breathed oronasally even at rest. In contrast, 5 subjects (17 percent, all females) breathed solely through the nose both at rest and throughout the exercise period, and one subject's nose/mouth breathing pattern was unpredictable and inconsistent. The percentage of subjects Niinimaa et al. (1980) observed breathing oronasally even at rest is in good agreement with the results of previous studies by Uddströmer (1940) and Saibene et al. (1978), although the 17 percent incidence for nose breathing involving all females is approximately double that found for males in studies by Uddströmer (1940) and Saibene et al. (1978). The fact that Niinimaa et al. (1980) did not observe any male nose breathers is probably a reflection of the small number of subjects studied.

With any of the common obstructive forms of nasal pathology such as allergic, viral, or vasomotor rhinitis or septal deviation, the proportion of ventilation passing through the mouth is higher at rest and at any level of exercise. Healthy young adults without nasal pathology, who breathe predominantly through the nose at rest, shift to breathing through the nose and mouth when minute ventilation is approximately 35 l/min (Niinimaa et al., 1980; Saibene et al., 1978). Niinimaa et al. (1981) found that during exercise requiring a rate of ventilation of 35 l/min, 57 percent of the air passed nasally. The onset of oronasal breathing did not show a significant sex difference and was quite consistent individually. Between individuals, however, the variation was considerable, as reflected by the large standard deviation of 10.8 l/min (Niinimaa et al., 1980). Camner and Bakke (1980) studied the proportion between the air inhaled through the nose and mouth during two common activities, conversation and reading a newspaper, where the breathing is calm. They found that while conversing casually all subjects inhaled less air via the nose than while reading with the mouth closed: on the average less than half the volume. Studies on subjects breathing through a mouthpiece at rest (minute ventilation of 6 to 8 l/min) provide conservative estimates of the mass burden of particles, since the total quantity of ventilation passing through the mouth is significantly less than that which would pass through the mouth in the same subjects breathing freely through the nose and mouth while performing enough exercise to require a minute ventilation of 35 l/min. A minute ventilation of 35 l/min corresponds to anywhere from light to moderate exercise according to various sources (Zenz, 1975; Higgs et al., 1967; American Heart Association, 1973; Jones et al., 1975; Intermountain Thoracic Society, 1975; International Commission on Radiological Protection, 1975).

11.1.5 Mechanisms of Particle Deposition

The behavior of inhaled airborne particles in the respiratory airways and their alternative fate of either deposition or exhalation depend on aerosol mechanics under the given physiological and anatomical conditions (Yeh et al., 1976; DuBois and Rogers, 1968). This behavior is usually described in terms of nonreactive stable spherical particles whose physical properties do not vary during the breathing cycle. Behavior of hygroscopic and deliquescent particles is more complex.

Figure 11-2 illustrates the five primary physical processes that lead to aerosol particle contact with the wall of the airways. Contact of particles with moist airway walls results in attachment and irreversible removal of the particle from the airstream. The contact process can occur during inspiration or expiration of a single breath or subsequently if a particle has been transferred to unexhaled lung air (Engel et al., 1973; Davies, 1972; Altshuler, 1961).

Electrostatic attraction of particles to the walls of the respiratory airways is probably a minor mechanism of deposition in most circumstances. Pavlik (1967) predicted that light air ions (which would include some atmospheric aerosol nuclei) would be deposited by electrostatic attraction in the mouth and throat and suggested that the tonsils were naturally charged for this purpose. Fraser (1966) found that an average of 1000 electronic units of charge per aerosol particle, a very large charge that does not normally occur, doubled the inhalation deposition in experimental animals. Melandri et al. (1977) reported enhanced deposition of inhaled monodisperse aerosols in humans when the particles were charged. Longley (1960) and Longley and Berry (1961) found the charge of the subject to have an influence on deposition. Similar observations have been made in *in vitro* studies (Chan et al., 1978). The airways are covered by a relatively conductive electrolytic liquid that probably precludes the buildup of

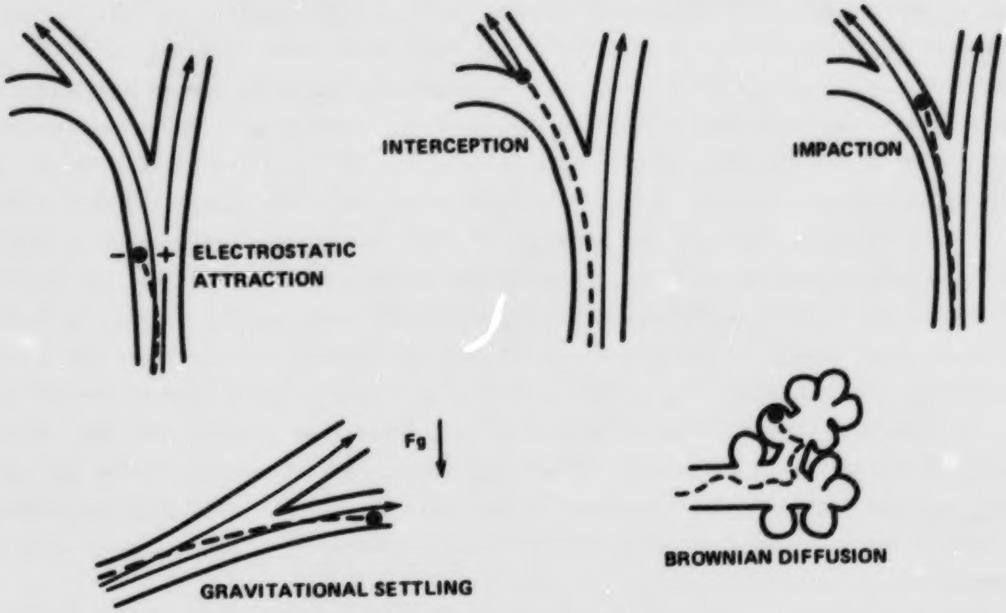


Figure 11-2. Representation of five major mechanisms of deposition of inhaled airborne particles in the respiratory tract.

Source: Raabe (1979).

forceful electric fields. Charged particles are therefore collected primarily by image charging as they near the wall of an airway or by mutual repulsion from a unipolarly charged cloud with a high concentration of particles (Yu, 1977). The role of this mechanism depends on particle source, concentration, age, and special electrical phenomena in the environment, as well as the residence time of the aerosol in the airways. This mechanism can be expected to have a small role, if any, in the deposition of atmospheric environmental aerosols.

Interception consists of noninertial incidental meeting of a particle and the lining of the airway and thus depends on the physical size of the particle. It is important primarily for particles with large aspect ratios, such as long fibrous particles of asbestos (Harris and Fraser, 1976). Interception may be expected to play a minor role in the inhalation deposition of most environmental aerosols.

Impaction dominates deposition of particles larger than 3 μm D_{ae} in the nasopharyngeal and tracheobronchial regions (Pattle, 1961a; Bohning et al., 1975). In this process, changes in airstream direction or magnitude of air velocity streamlines or eddy components are not followed by airborne particles because of their inertia. For example, if air is directed toward an airway surface (such as a branch carina) but the forward velocity is suddenly reduced because of change in flow direction, inertial momentum may carry larger particles across the air streamlines and onto the surface of the airway. Impaction at an airway branch has been likened to impaction at the bend of a tube, providing theoretical estimates of the impaction probability (Johnston and Muir, 1973; Yeh, 1974; Cheng and Wang, 1975) and was studied in a bifurcating tube model by Johnston and Schroter (1979). Aerodynamic separation of this type is characterized satisfactorily in terms of the particle aerodynamic diameter. The airflow in the trachea and major bronchi in humans is turbulent and disturbed by the larynx, so that turbulent impaction plays a role in deposition in these larger airways (Schlesinger and Lippmann, 1976). Breathing patterns involving higher volumetric flowrates would tend to impact smaller particles. In contrast, the passages of the nose contain smaller airways, and the convective mixing spaces of the nasal turbinates would be expected to collect some particles as small as 1 or 2 μm D_{ae} by impaction. Hence, impaction is an important process affecting the inhalation deposition in the human airways of environmental aerosol particles greater than 1 μm in aerodynamic diameter.

Gravitational settling occurs because of the influence of the earth's gravity on airborne particles. Deposition of particles by this mechanism can occur in all airways except those very few that are vertical. The probability of gravitational deposition is usually estimated with equations describing gravitational settling of particles in an inclined cylindrical tube under laminar flow conditions (Wang, 1975; Heyder and Gebhart, 1977). This deposition depends on the residence time and particle concentration distribution in the airway segments, the angle of incline of the segment with respect to gravity, and the aerodynamic diameter of the particle. Deposition by gravitational settling is therefore characterized in terms of the particle aerodynamic diameter. This mechanism has an important influence on the deposition of particles larger than 0.5 μm D_{ae}. Settling has an important role in the deposition of environmental aerosols in the distal region of the bronchial airways and in the pulmonary region.

Deposition by diffusion results from the random (Brownian) motion of very small particles caused by bombardment of the gas molecules in air. The magnitude of this motion can be described by the diffusion coefficient for a given physical particle diameter. Since particles larger than 0.5 μm D_{ae} have relatively small diffusional mobility compared with sedimentation or inertia, diffusion primarily affects deposition of particles with physical diameters smaller than 0.5 μm D_{ae}. For particles of 0.5 μm D_{ae} with a physical density of about 1 g/cm^3, the influences of inertial properties and diffusional properties on lung deposition are about equal. Accurate calculation of the diffusional deposition of aerosols in the airways requires information concerning the three-dimensional velocity profile of airflow in each airway segment. If the flow of a given segment is laminar and approximately Poiseuille, the probability of deposition by diffusion might be approximated using the Gormley and Kennedy (1949) equation for a cylindrical pipe. This assumes, however, the aerosol is mixed at the entrance of the cylinder and that the flow is constant. It therefore overestimates deposition in lung segments where there is minimal mixing between tidal and residual air and reversible laminar flow between segments.

The diffusivity and interception potential of a particle depend on its physical size, whereas the inertial properties of settling and impaction depend on its aerodynamic diameter. These two measures of size may be quite different, depending on particle shape and physical density. Because the main mechanism of deposition is diffusion for particles whose physical (geometric) size is less than 0.5 μm D and impaction and settling above 0.5 μm D_{ae}, 0.5 μm is convenient to use as the boundary. Although this convention may lead to confusion in the case of very dense particles, most environmental aerosols have densities below 3 g/cm^3, and the deposition probability tends to have a minimum plateau between 0.2 μm and 1 μm D_{ae}.

Information concerning breathing patterns and respiratory physiology, the anatomical and geometrical characteristics of the airways, and the physical behavior of insoluble spherical particles can be used to develop theoretical models of regional deposition (Findeisen, 1935; Landahl et al., 1951; Landahl, 1963; Beeckmans, 1965; Yu, 1978). In these models, deposition of inhaled aerosols in a given region of the respiratory tract or in the entire tract is expressed as a fraction of inhaled particles. Deposition fraction is the ratio of the number or mass of particles deposited in the respiratory tract to the number or mass of particles inhaled. The undeposited fraction represents those particles that are exhaled after inhalation. For example, pulmonary deposition is the ratio of the number or mass of particles deposited in the unciliated small airways and gas exchange spaces of the parenchyma of the lung to the number or mass of particles entering the nose or mouth. The fraction not deposited in the pulmonary region is either deposited in some other region or exhaled. Similarly, deposition fractions can be defined for the other regions of the respiratory tract.

Most model calculations treat the various mechanisms of deposition as independently occurring phenomena. Such processes as Brownian diffusion and gravitational settling, however, will interfere with each other when their effects are of comparable magnitude, and that interference can reduce the combined deposition to less than the sum of the separate depositions (Goldberg et al., 1978). Taulbee and Yu (1975) developed a theoretical deposition model that allows for the combined effects of the primary deposition mechanisms and features an imaginary expanding tube model of the airway system (Weibel, 1963) based on cross-sectional areas and airway lengths.

Historically, the most widely used models of regional deposition versus particle size were developed by the ICRP under the chairmanship of P. E. Morrow (Morrow et al., 1966). These models were developed to determine radiation exposure from inhaled radioactive aerosols. Although the ICRP aerosol deposition and clearance models were not intended for broad application to environmental aerosols, they have been so applied by some scientists. The ICRP Task Group used the anatomical model and impaction and sedimentation equations of Findeisen (1935) and the general methods of Landahl (1950, 1963) for calculating deposition in the tracheobronchial and pulmonary regions. The Gormley and Kennedy (1949) equation for cylindrical tubes was used for calculating diffusional deposition. For head deposition, inhalation through the nose with a deposition efficiency given by the empirical equation of Pattle (1961a) was used. Particles were assumed to be insoluble, stable, and spherical with physical densities of 1 g/cm^3 and log-normally distributed with a σ_g as high as 4.5. When the results were expressed in terms of MMD for these various sized distributions of unit density (equivalent to the MMAD), the range of the expected regional deposition values was relatively narrow.

At the time the ICRP Task Group models were developed, the available human data were primarily total deposition values for polydisperse and sometimes unstable aerosols (Landahl and Herrmann, 1948; Davies, 1964b; Van Wijk and Patterson, 1940; Brown et al., 1950; Dautrebande and Walkenhurst, 1966; Morrow et al., 1958; Landahl and Black, 1947). Since then, the deposition in humans of monodisperse insoluble, stable aerosols of different sizes has been measured under different breathing conditions. Extensive studies were conducted by Lippmann (1977), Heyder et al. (1975, 1980a,b), Stahlhofen et al. (1980), Chan and Lippmann (1980), and Giacomelli-Maltoni et al. (1972). Additional useful data were reported by Palmes and Wang (1971), Shanty (1974), George and Breslin (1967), Altshuler et al. (1967), Hounam et al. (1971a,b), Foord et al. (1976), Pavia et al. (1977), among others (Muir and Davies, 1967; Taulbee et al., 1978; Hounam, 1971; Heyder, 1971; Heyder and Davies, 1971; Fry and Black, 1973).

11.2 DEPOSITION IN MAN AND EXPERIMENTAL ANIMALS

11.2.1 Insoluble and Hydrophobic Solid Particles

11.2.1.1 Total Deposition--The background information in Section 11.1 demonstrates that a knowledge of where particles of different sizes deposit in the respiratory tract and the extent of their deposition is necessary for understanding and interpreting the health effects associated with exposure to particles and SO_2. As was seen, the respiratory tract can be divided into regions on the basis of structure, size, and function. Insoluble particles depositing in the various regions contact or affect different cell populations and have large differences in retention times and clearance pathways (see Section 11.3).

If the quantity of aerosol exhaled is compared with that inhaled, the data can be expressed as total deposition, but regional involvement cannot be distinguished. By tagging the test aerosols with radiolabels, investigators have been able to separate deposition by region, beginning with either nasal and nasopharyngeal deposition for nose breathing or oral and pharyngeal deposition for mouth breathing. The measurement of clearance of the radiolabeled aerosol from the thorax can be used to separate early clearance, indicative of tracheobronchial (TB) deposition, from more slowly cleared pulmonary (P) deposition.

Total respiratory tract deposition with nose breathing is given in Figure 11-3, and total deposition with mouth breathing is depicted in Figure 11-4. Analyses based on the difference between concentrations of inhaled and exhaled particles, as well as those based on external *in vivo* measurements of radiolabeled particles, are represented in the studies from which these figures are taken. With nose breathing, complete deposition can be expected for particles larger than about 4 µm D_{ae}. Mouth breathing bypasses much of the filtration capabilities of the extrathoracic (ET) region, and a shift upward to around 10 µm D_{ae} occurs before there is complete deposition of the inhaled particles.

The various studies all appear to show the same trend. The particle size for minimum deposition is less clear for nasal breathing than for mouth breathing, for which minimum deposition is at about 0.5 µm D_{ae}. Heyder and coworkers (1973a, 1973b, 1975) carefully matched breathing patterns in subjects in their studies of the deposition of 0.5 µm D_{ae} particles on which there were no electrical charges; their data are the deposition minima in Figure 11-4. Thus far, deposition of particles less than 0.1 µm diameter was studied in human subjects only by Swift et al. (1977).

Heyder and coworkers (1975, 1980a,b) studied the effects of respiratory parameters on aerosol deposition in systematic experiments comparing deposition of different sized monodisperse aerosols in human volunteers at different tidal volumes, flowrates, and breathing frequencies. For particles between 0.1 and 4.0 µm in diameter, Heyder et al. (1975) measured total respiratory deposition during either nose or mouth breathing while sequentially maintaining a given TV, breathing rate, or inspiratory flowrate and then varying the other two parameters. They demonstrated several important features of aerosol deposition in the human respiratory airways. Heyder et al. (1980a,b) extended these studies to particles as large as 9 µm D_{ae}; in the mouth-breathing experiments they also determined alveolar deposition.

With volumetric flowrate held at 15 l/min while the subject breathed through the mouth, the particle size yielding the lowest deposition changed from 0.66 µm D_{ae} at TV 250 ml to 0.46 µm

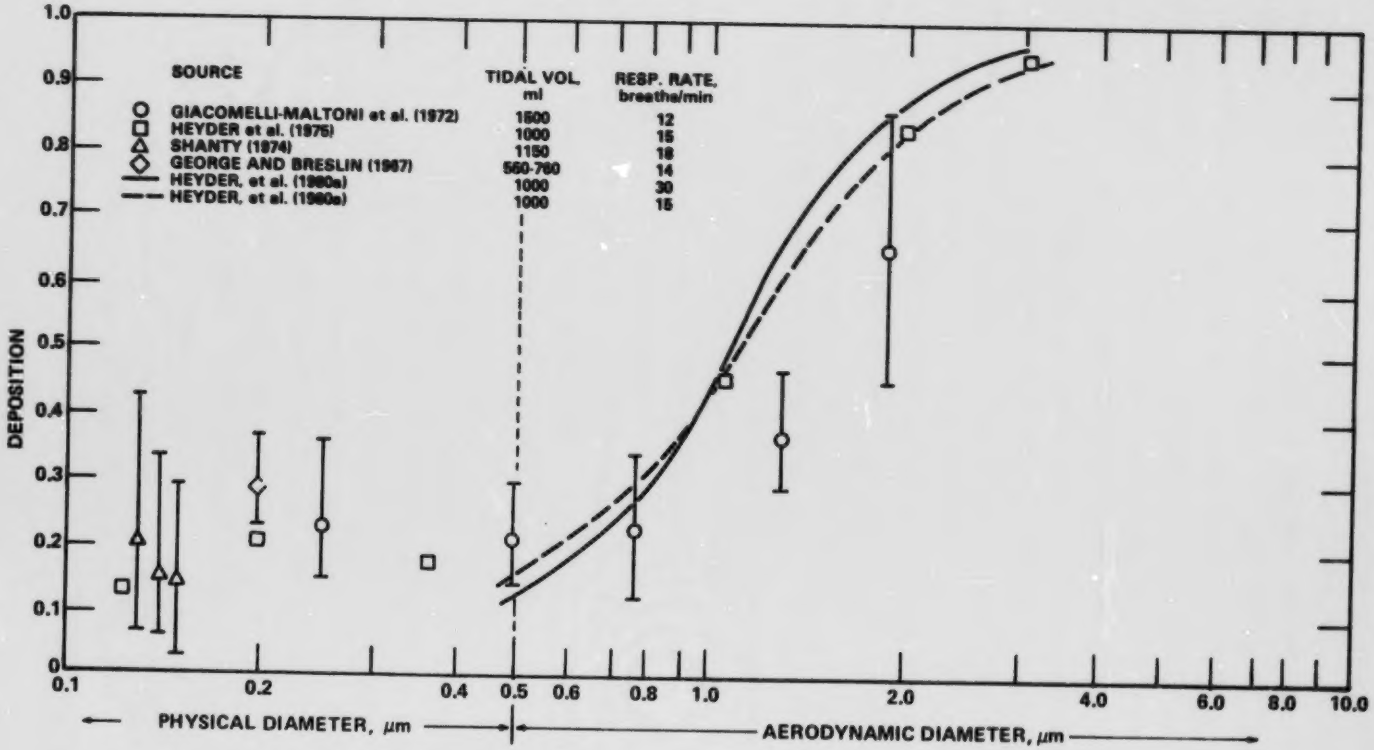


Figure 11-3. Deposition of monodisperse aerosols in the total respiratory tract for nasal breathing in humans as a function of aerodynamic diameter, except below 0.5 μm, where deposition is plotted vs. physical diameter. The data are individual observations, averages, and ranges as cited by the various investigators.

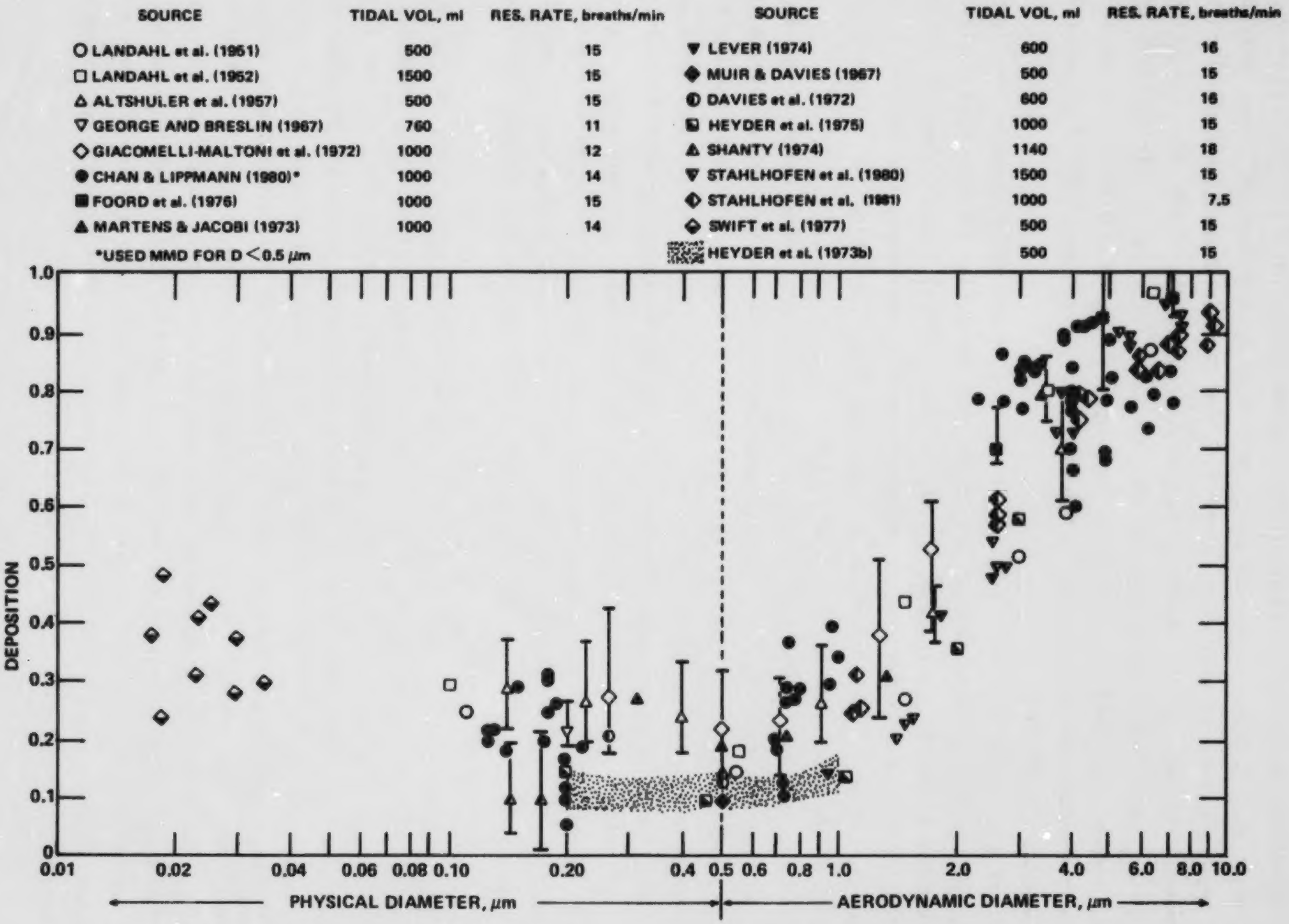


Figure 11-4. Deposition of monodisperse aerosols in the total respiratory tract for mouth breathing in humans as a function of aerodynamic diameter, except below 0.5 μm, where deposition is plotted vs. physical diameter. The data are individual observations, averages, and ranges as cited by the various investigators.

at TV 2000 ml. Breathing at TV 1000 ml changed this minimum deposition size from 0.58 µm D_{ae} at 30 BPM to 0.46 µm at 3.75 BPM. Hence, the particle size of minimum deposition was reduced with increased residence time of particles in the lung and the net deposition for all particles was increased. In fact, as the breathing rate went from 3.75 to 30 BPM, the deposition fraction at 1 µm D_{ae} went from 0.08 to 0.4, an increase of a factor of 5. In contrast to mouth breathing, however, the particle size of minimum deposition with nose breathing was independent of the residence time of particles in the respiratory tract when 1000 ml of aerosol at different flowrates was inhaled.

When Heyder et al. (1975) kept the breathing frequency constant while changing the flowrate and having the subjects breathe through the mouth, the deposition for particles smaller than 1 µm D_{ae} remained essentially unchanged, indicating that inertial impaction was of little importance in the deposition of submicrometer aerosols. On the other hand, the deposition of particles larger than 1 µm D_{ae} was enhanced at high flowrates, indicating the influence of inertial impaction on the deposition of larger particles.

Sedimentation and impaction are competing deposition mechanisms, being governed by mean residence time and flowrate, respectively. Hence, impaction will be the dominant mechanism at high flowrates and short residence times, and most particles will be deposited by sedimentation at low flowrates and long residence times. Heyder et al. (1980a) showed that for 1 µm D_{ae} particles, total deposition for mouth breathing at 1000 ml TV increased with increasing mean residence time, indicating these particles were mainly deposited by sedimentation. For 8 µm D_{ae} particles, increasing flowrate increased deposition so that these particles were mainly deposited by impaction. A transition region was observed for particles about 4 µm D_{ae}; Heyder and coworkers (1980a) noted the transition region was shifted towards smaller particles for nose breathing.

11.2.1.2 Extrathoracic Deposition--The fraction of inhaled aerosol depositing in the ET region can be quite variable, depending on particle size, flowrate, and breathing frequency, and whether breathing is through the nose or through the mouth. During exertion, the flow resistance of the nasal passages cause a shift to mouth breathing in almost all individuals, thereby bypassing much of the filtration capabilities of the head and leading to increased deposition in the TB region. Extrathoracic deposition is shown for nose breathing in Figure 11-5 and for mouth breathing in Figure 11-6. Deposition in this region is usually plotted as a function of D_{ae}^2Q, since this is a convenient parameter for normalizing impaction-dominated deposition data when the actual flowrates are not identical (Pattle, 1961a; Stahlhofen et al., 1980; National Academy of Sciences, 1980; Hounam et al., 1969, 1971a). For reference, a scale showing aerodynamic diameter when Q = 30 l/min is also shown, since this flowrate approximates the average flowrate for the studies cited in these figures.

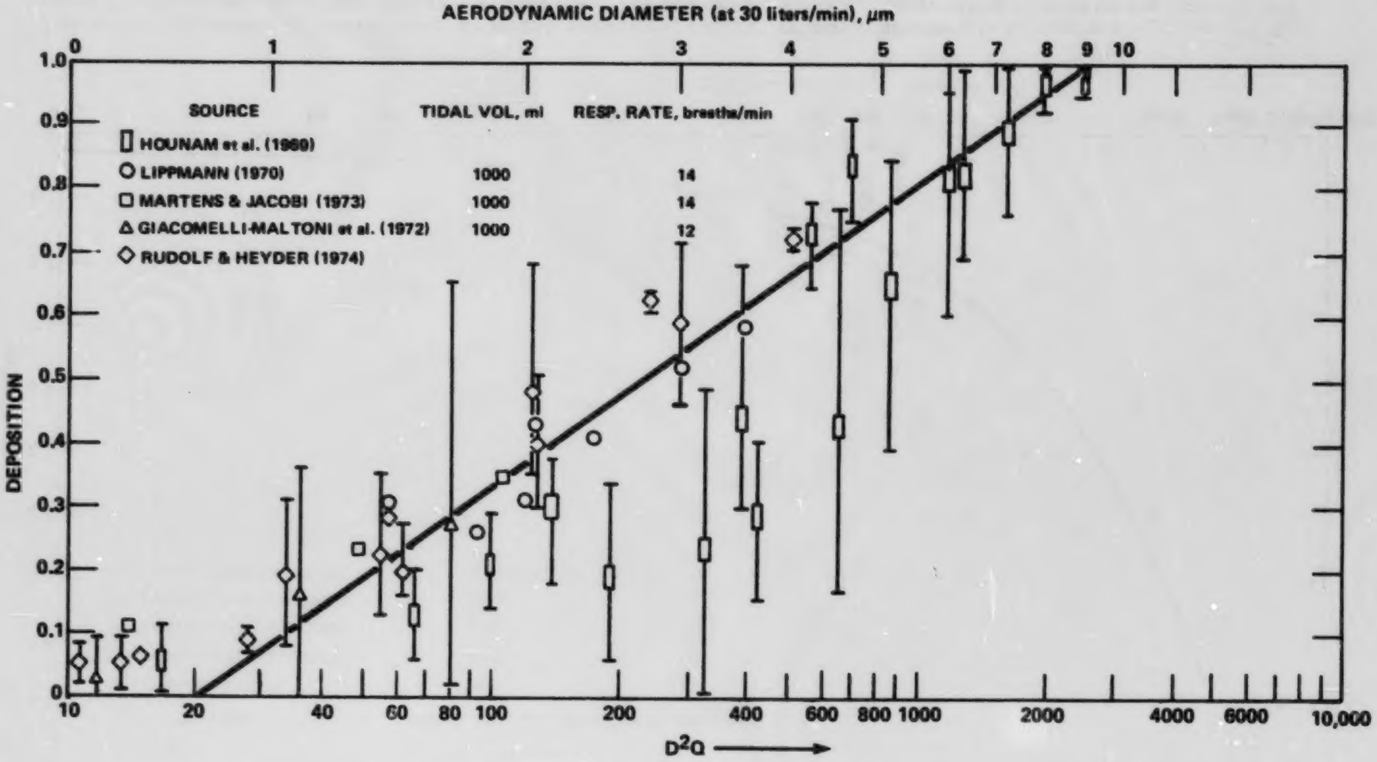


Figure 11-5. Deposition of monodisperse aerosols in the extrathoracic region for nasal breathing in humans as a function of D^2Q, where Q is the average inspiratory flow rate in liters/min. The solid line is ICRP deposition model based on the data of Pattle (1961a). Other data show the median and range of the observations as cited by the various investigators.

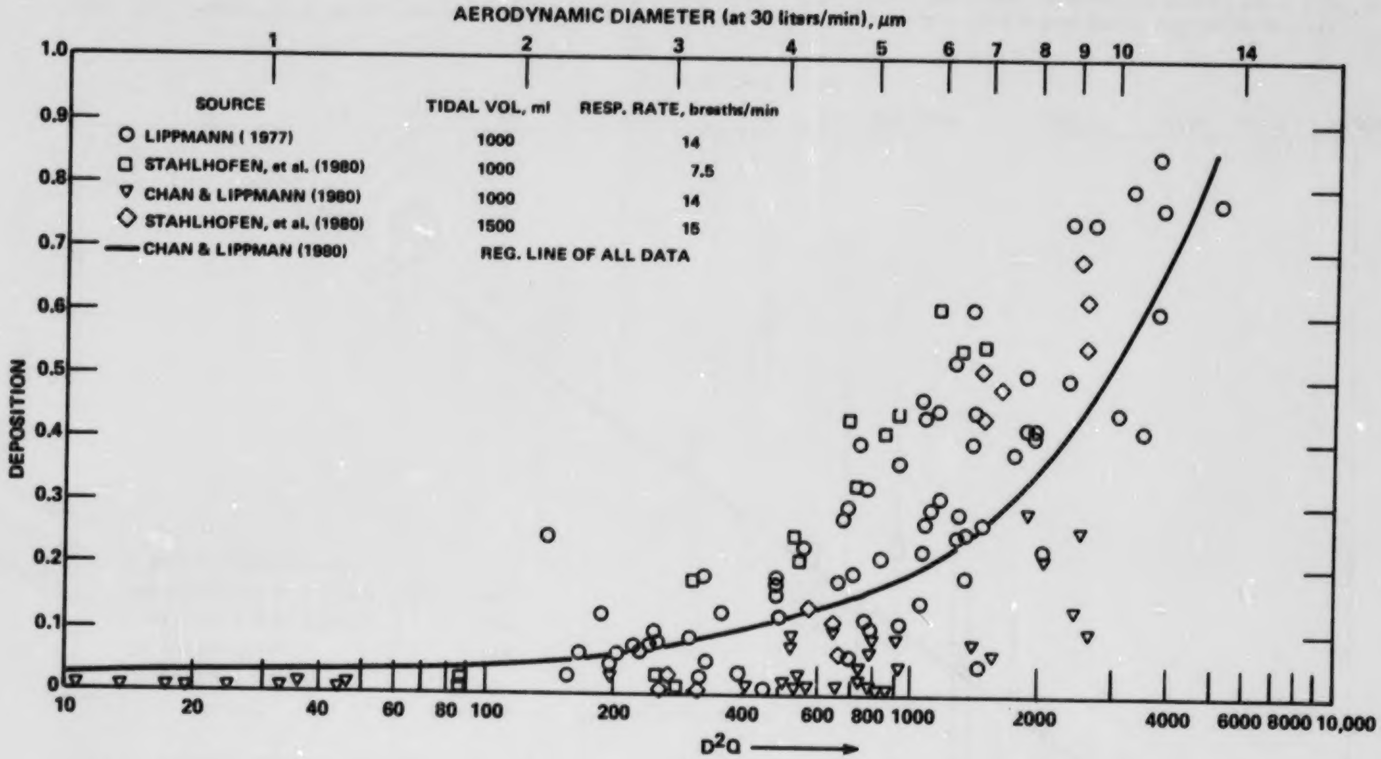


Figure 11-6. Deposition of monodisperse aerosols in extrathoracic region for mouth breathing in humans as a function of D^2Q, where Q is the average inspiratory flow rate in liters/min. The data are the individual observations as cited by the various investigators. The solid line is the overall regression derived by Chan and Lippmann (1980).

Particles larger than about 10 μm D_{ae} entering the nose are effectively deposited in the ET region (Figure 11-5). Also, deposition is slight (10 percent) for particles less than 1 μm D_{ae}. Similarly, for 10 and 1 μm D_{ae} particles under conditions of mouth breathing (Figure 11-6), ET deposition is about 65 and 2 percent, respectively. The regression curve shown in Figure 11-6 is from Chan and Lippmann (1980) who used their own data, as well as the data of Lippmann (1977) and Stahlhofen et al. (1980) for Q = 45 liter/min, in their analysis. As indicated by Chan and Lippmann (1980), some of the lower values of ET deposition may be due to partial clearance to the stomach before the measurement of head deposition was obtained. Particles can be swallowed even when the subject consciously tries to avoid swallowing (Lippmann, 1977; Stahlhofen et al., 1980).

11.2.1.3 Tracheobronchial Deposition--As was seen earlier, when aerosols are inhaled through the nose, relatively efficient ET filtration eliminates the passage of most particles larger than about 10 μm D_{ae} to the TB region. Mouth breathing markedly alters the deposition of inhaled particles in humans, in that larger particles can enter both the TB and P regions to a greater extent (Morrow et al., 1966; Lippmann, 1977; Heyder et al., 1980a,b; Stahlhofen et al., 1980). For mouth breathing, TB deposition expressed as a fraction of the particles entering the trachea is shown in Figure 11-7 plotted against particle size. Approximately 80 to 90 percent of 8 to 10 μm D_{ae} particles entering the trachea are deposited in the TB region, as compared with less than 10 percent for particles less than 1 μm D_{ae}. The increased penetration of large particles deeper into the respiratory tract when a person breathes through the mouth can be seen from the 20 to 30 percent experimental TB deposition data for particles 8 to 10 μm D_{ae} (Stahlhofen et al., 1980). The solid curve in Figure 11-7 is from Chan and Lippmann (1980), depicting the experimental TB deposition data from their investigations using the average value of a new anatomic parameter, the bronchial deposition size, for the average Q value measured in their study (Q = 39 l/min). This parameter enables the classification of various individuals and populations according to their TB deposition efficiencies and is an improvement over the characteristic airway dimension parameter developed earlier by Palmes and Lippmann (1977).

Deposition in the TB region is influenced by both impaction and sedimentation, with the relative contribution of these two mechanisms changing with particle size and airflow rate. Impaction predominates for deposition of particles larger than about 3 μm D_{ae} and flowrates greater than about 20 l/min; on the other hand, sedimentation deposition becomes a larger fraction of a diminishing TB component for smaller particles and lower flows (Lippmann, 1977). The importance of impaction for TB deposition is reflected by deposition often being plotted against the inertial parameter, D_{ae}^2Q (Stahlhofen et al., 1980 and Lippmann, 1977).

For a given particle size, TB deposition with mouth breathing varies greatly from subject to subject among nonsmokers, cigarette smokers, and patients with lung disease (Lippmann et al.,

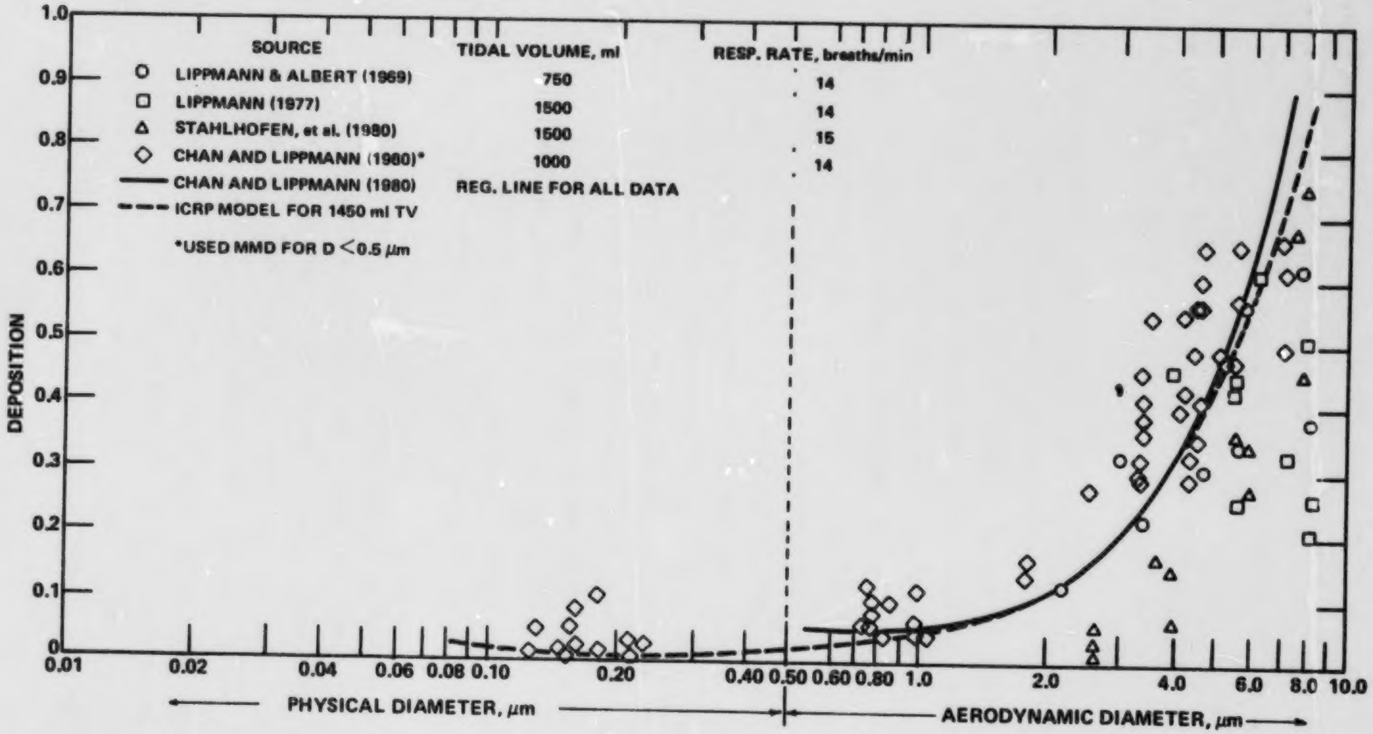


Figure 11-7. Deposition of monodisperse aerosols in the tracheobronchial region for mouth breathing in humans in percent of the aerosols entering the trachea as a function of aerodynamic diameter, except below 0.5 μm, where deposition is plotted vs. physical diameter as cited by different investigators. Dashed line is ICRP model for 1450 ml tidal volume. The solid line is the overall regression derived by Chan and Lippmann (1980).

1971). On the average, TB deposition is slightly elevated in smokers and greatly elevated in patients with lung disease (Lippmann et al., 1977; Cohen, 1977). Each subject, however, exhibits a characteristic and reproducible relationship between particle size and deposition as indicated by the data of Stahlhofen et al. (1980), depicted in Figure 11-8. For the two breathing patterns shown, the steep increase of the ET deposition values with increasing particle size is accompanied by a corresponding decrease in TB deposition, so that TB deposition, as a function of particle aërodynamic diameter, may be described by a bell-shaped curve with a maximum (Stahlhofen et al., 1980). Although these investigators did not experimentally study particles larger than 9 μm D_{ae}, extension of their bell-shaped curves would support the conclusion of Miller et al. (1979) that about 10 percent of particles as large as 15 μm D_{ae} can enter the TB region during mouth breathing. Miller et al. (1979) used the TB deposition data of Lippmann (1977) and aerodynamic diameters computed at a mean flowrate of 30 l/ min. This flowrate is bracketed by the mean flowrates of 15 and 45 l/min used by Stahlhofen et al. (1980).

The data of Stahlhofen et al. (1980) in Figure 11-8 on three subjects show lower values and less scatter than the other data contained in the figure. Chan and Lippmann (1980) cited two possible explanations for the differences. Stahlhofen and coworkers (1980) used constant respiratory flowrates in comparison with the variable flowrates used by Chan and Lippmann (1980). Also, the two laboratories used different bases to separate the initial thoracic burden into TB and P components. Stahlhofen et al. (1980) extrapolated the thoracic retention values measured during the week after the end of bronchial clearance back to the time of inhalation; they considered P deposition to be the intercept at that time, with the remainder of the thoracic burden considered as TB deposition. This approach yields results similar to, but not identical with, those obtained by treating TB deposition as equivalent to the particles cleared within the first day.

Deposition calculations usually group lung regions without regard to nonuniformity of the pattern of deposited particles within the regions. Schlesinger and Lippmann (1978) found that nonuniform deposition in the trachea could be caused by the airflow disturbance of the larynx. Bell and Friedlander (1973) and Bell (1978) observed and quantified particle deposition as it occurs at a single airway bifurcation and found it to be highly nonuniform and heaviest around the carinal arch. Raabe et al. (1977) observed that the relative lobar pulmonary deposition of monodisperse aerosols was up to 60 percent higher in the right apical lobes than in others of small rodents (corresponding to the human right upper lobe) and that the difference was greater for 3.05 and 2.19 μm D_{ae} particles than for smaller particles. In addition, Raabe et al. (1977) showed that these differences in relative lobar deposition were related to the geometric mean number of airway bifurcations between trachea and terminal bronchioles in each lobe for rats and hamsters. Since similar morphologic differences occur in human lungs, nonuniform

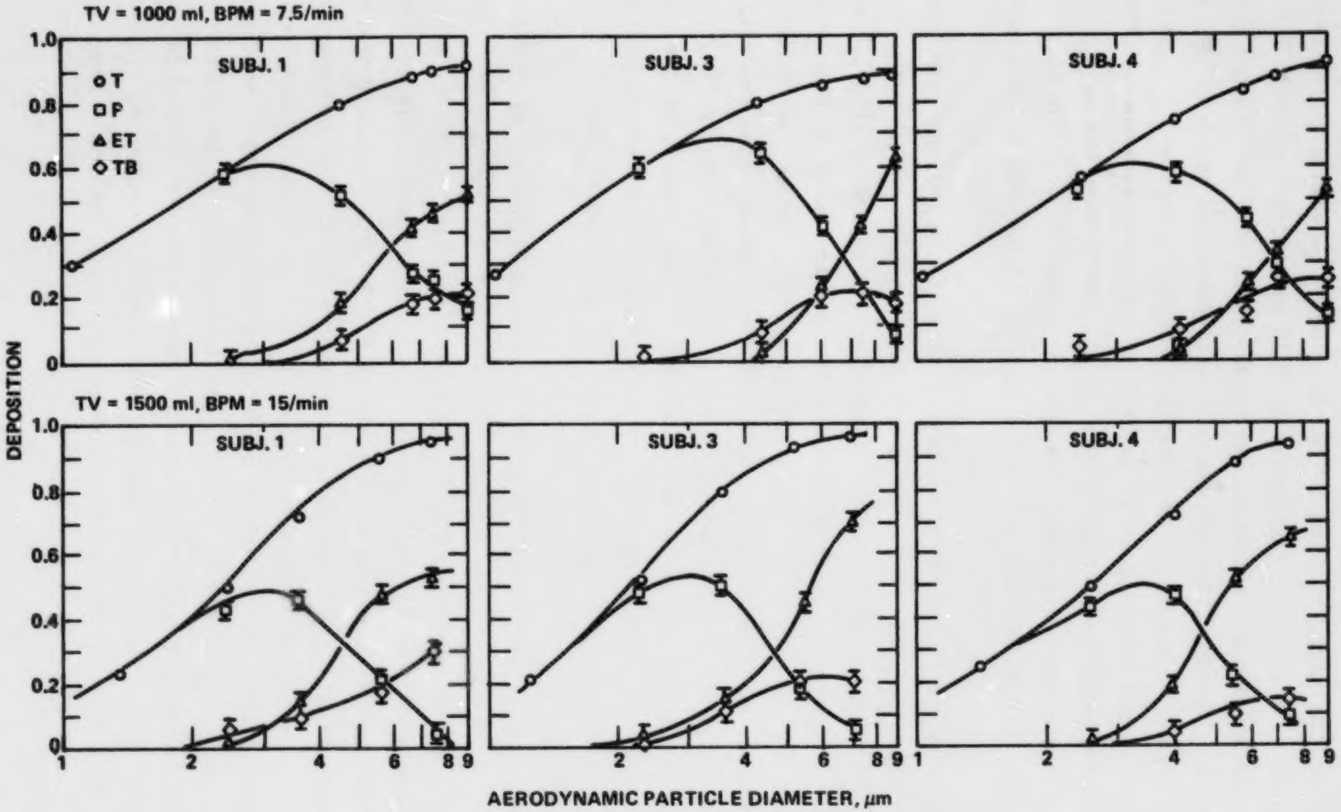


Figure 11-8. Total and regional depositions of monodisperse aerosols with mouth breathing as a function of the aerodynamic diameter for three individual subjects as cited by Stahlhofen et al. (1980). (T = Total, TB = Tracheobronchial, P = Pulmonary, ET = Extrathoracic, TV = Tidal Volume, BPM = Breaths Per Minute.)

lobar deposition should also occur. Schlesinger and Lippman (1978) found nonuniform deposition in the lobar branches of a hollow model of the TB airways with enhanced carinal deposition and were able to demonstrate a correlation of higher lobar deposition and the reported incidence of bronchogenic carcinoma in the different human lobar bronchi. Occupational lung diseases, such as silicosis and asbestosis, also show distinctive distributional features (Morgan and Seaton, 1975).

11.2.1.4 Pulmonary Deposition--Pulmonary deposition as a function of particle size is shown in Figure 11-9. All of the experimental points plotted were obtained in mouth-breathing studies on nonsmoking normal subjects who inhaled monodisperse aerosols.

The eye-fit band approximately encompasses the range of deposition values obtained in the studies cited; a variety of TV's and breathing frequencies were used. Also shown in Figure 11-9 are the deposition curve from the predictive model of Yu (1978) and an estimate of the P deposition that could be expected for nose breathing (Lippmann, 1977). Lippmann (1977) derived the estimate by analysis of the difference in head retention during nose breathing and mouth breathing.

The P deposition curve peaks at about 3.5 µm D_{ae} with the middle of the eye-fit band in Figure 11-9 being located at about 50 percent deposition. However, the data of Stahlhofen et al. (1980) for a tidal volume of 1000 ml and 7.5 BPM (reflective of breathing very slowly and deeply) showed that P deposition of 3.5 µm D_{ae} particles can be as high as 70 percent.

For nose breathing, the size associated with maximum deposition shifts downward to about 2.5 µm D_{ae}. Also, the deposition peak is much less pronounced (about 25 percent), with a nearly constant P deposition of about 20 percent for all sizes between 0.1 µm and 4 µm D_{ae}.

Pulmonary and total deposition of Fe_2O_3 (density 3.2 g/cm^3) particles and di-2-ethylhexyl sebacate droplets for mouth breathing was evaluated by Heyder et al. (1980a,b) as a function of aerodynamic diameter for two breathing patterns. Some results with di-2-ethylhexyl sebacate particles were reported by Heyder et al. (1980a) in terms of particle diameter. They are presented here for uniformity in terms of aerodynamic diameter, since these particles were close to unit density. Keeping the mean volumetric flowrate constant at 250 ml/s and allowing the mean residence time to vary between 2 and 8 s, they observed that as the mean residence time increased, the particle size having the greatest probability of deposition decreased. With this mean flowrate, particles smaller than about 2.4 µm D_{ae} were exclusively deposited in the P region, indicating that their inertia was not sufficiently high for impaction loses. When the mean flowrate was increased to 750 ml/s and the mean residence time was 2 s, particles with an aerodynamic diameter smaller than about 1.5 µm were exclusively deposited in the P region of the respiratory tract. The data of Heyder et al. (1980a,b) also showed that the particle size associated with the peak of the deposition curve and the magnitude of the peak decrease as the mean flowrate increases. In the above studies, maximum P deposition was at 3.5 and 3 µm D_{ae} when Q was 15 and 45 l/min, respectively.

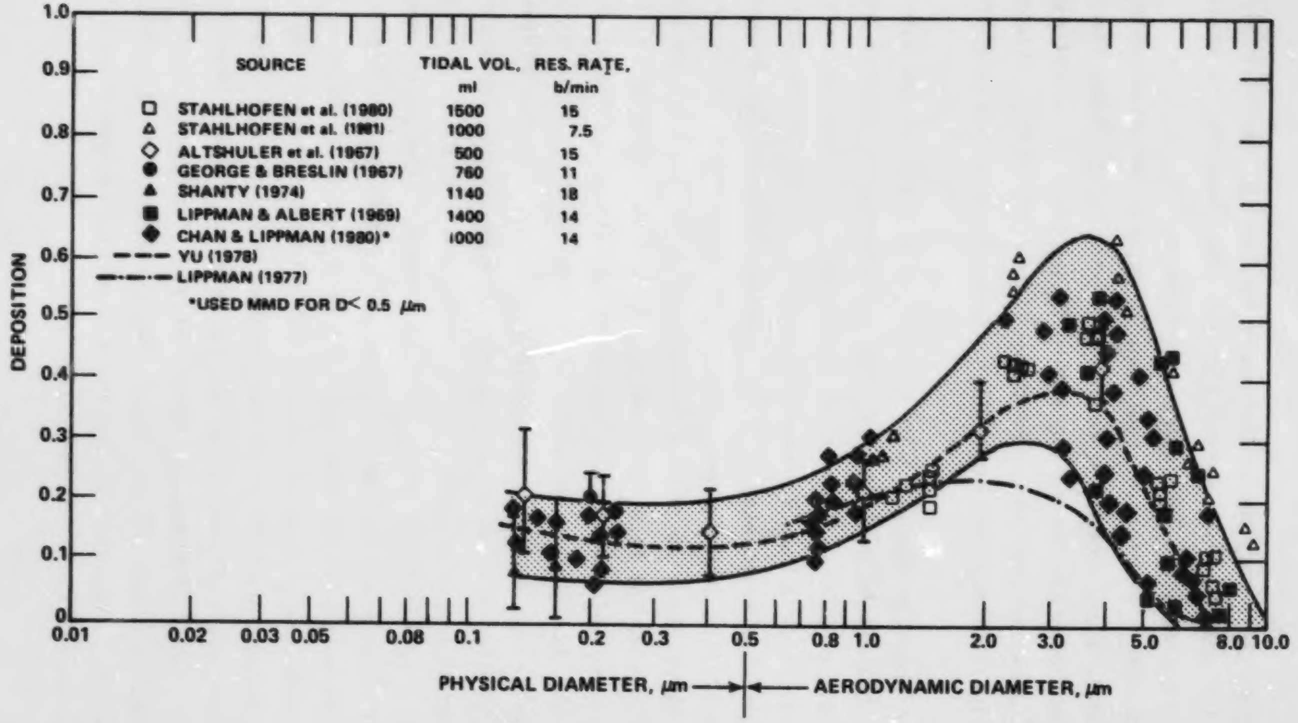


Figure 11-9. Deposition of monodisperse aerosols in the pulmonary region for mouth breathing in humans as a function of aerodynamic diameter, except below 0.5 μm, where deposition is plotted vs. physical diameter. The eye-fit band envelops deposition data cited by the different investigators. The dashed line is the theoretical deposition model of Yu (1978) and the broken line is an estimate of pulmonary deposition for nose breathing derived by Lippmann (1977).

11.2.1.5 Deposition in Experimental Animals--Since much information concerning inhalation toxicology is collected with beagles or rodents, the comparative regional deposition in these experimental animals must be considered to help interpret, from a dosimetric viewpoint, the possible implications for humans of animal toxicological results.

The study by Holma (1967) on rabbits examined mucociliary clearance rates, but Lippmann (1977) derived TB deposition information by further analyzing the data. Lung retention curves indicated that the TB deposition of 6 μm polystyrene spheres varied from 40 to 93 percent of the total lung deposition, with a median of 60 percent. A median of 29 percent was found for 3 μm particles. The above values are remarkably close to the available data for humans. Cuddihy et al. (1973) measured the regional deposition of polydisperse aerosols in beagles with TV about 170 ml at about 15 BPM and expressed the results as mass deposition percentage versus mass median aerodynamic resistance diameter ($MMAD_{ar}$) that ranged from 0.42 to 6.6 μm with geometric standard deviation $\sigma_g = 1.8$. These results are summarized in Figure 11-10 and, compared with the ICRP Task Group Values for humans with TV 1450 ml, integrated to account for a $\sigma_g = 1.8$. In comparison with the TB deposition of large particles in rabbits exposed to monodisperse aerosols for one test at 6.6 μm D_{ar}, the TB deposition in beagles was about 44 percent of the total lung deposition. With sizes between 2.5 and 3 μm D_{ar}, the TB deposition ranged from 5 to 39 percent, with a median deposition of 9 percent. The particle size for minimum P deposition was approximately 0.6 μm D_{ar}, with P deposition at this size ranging from about 12 to 35 percent and total deposition from about 18 to 55 percent.

Somewhat different results were obtained by Phalen and Morrow (1973) in dogs exposed to a silver metal aerosol of 0.5 μm D_{ae} with a $\sigma_g = 1.5$. Total deposition averaged 17 percent, with a range of 15 to 19 percent. In the Phalen and Morrow (1973) study, the dogs inhaled through a tracheal tube so that there was no head deposition; in the study of Cuddihy et al. (1973), head deposition varied from negligible to 5 percent for 0.5 μm D_{ae} particles. In experiments using donkeys (Albert el al., 1968, 1969; Spiegelman et al., 1968), eight animals were tested periodically with monodisperse 3 to 3.5 μm D_{ae} Fe_2O_3 aerosol. Tracheobronchial deposition averaged 50 to 70 percent of the total lung deposition, with a median of 54 percent.

Raabe et al. (1977) measured the regional deposition of 0.1 to 3.15 μm D_{ae} monodisperse aerosols in rats (TV about 2 ml, 70 BPM) and Syrian hamsters (TV about 0.8 ml at about 40 BPM). Their results are summarized in Figure 11-11. The P deposition of 1 to 3 μm D_{ae} particles is about 6 to 9 percent in rats and hamsters; deposition of these same size particles in humans varies from 21 to 24 percent for nose breathing and from 20 to 50 percent for mouth breathing. For particles smaller than 1 μm D_{ae}, differences in P deposition between humans and these animal species decrease. Tracheobronchial deposition of particles 5 μm D_{ae} is slight (~ 5 percent) in rodents due to very efficient removal of these particles in the head. In

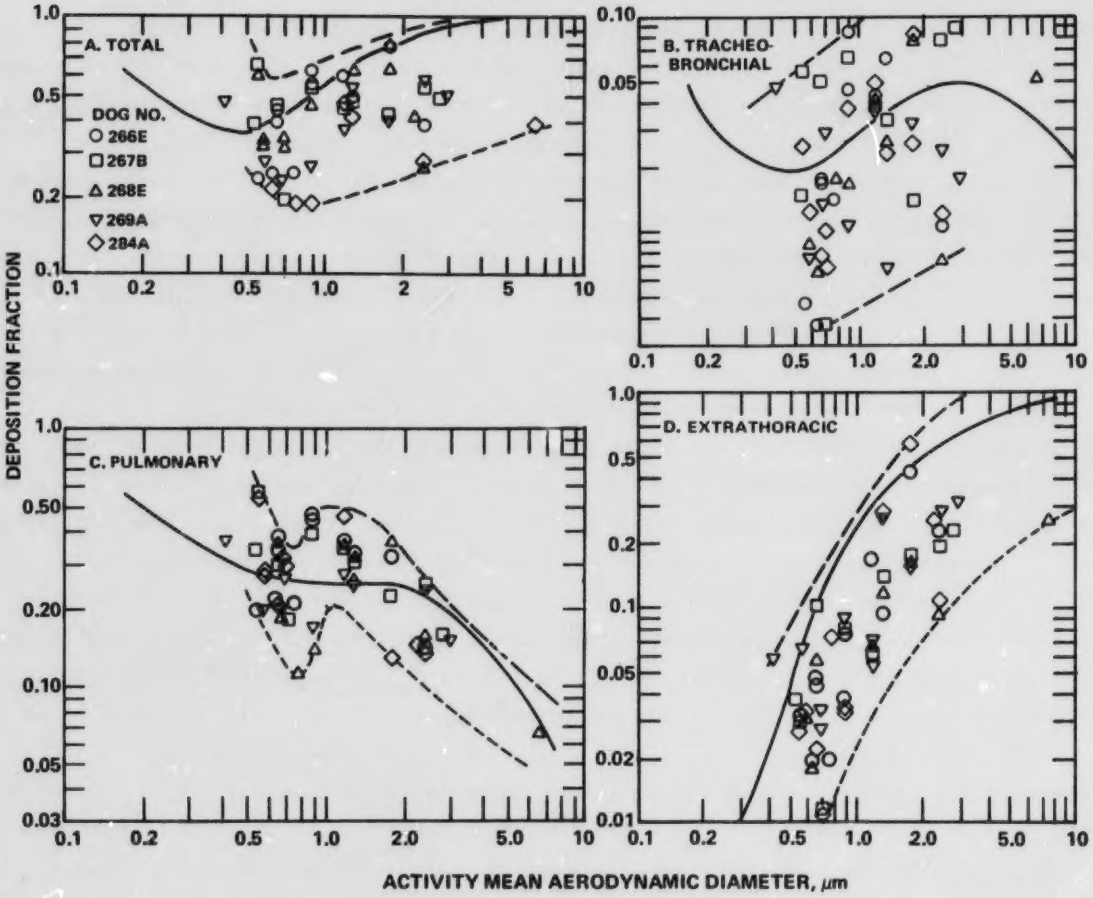


Figure 11-10. Deposition of inhaled polydisperse aerosols of lanthanum oxide (radio-labeled with ^{140}La) in beagle dogs exposed in a nose-only exposure apparatus showing the deposition fraction (A) total dog, (B) tracheobronchial region, (C) pulmonary aveolar region, and (D) extrathoracic region (adapted from Cuddihy et al. 1973). Dashed lines represent range of observed values.

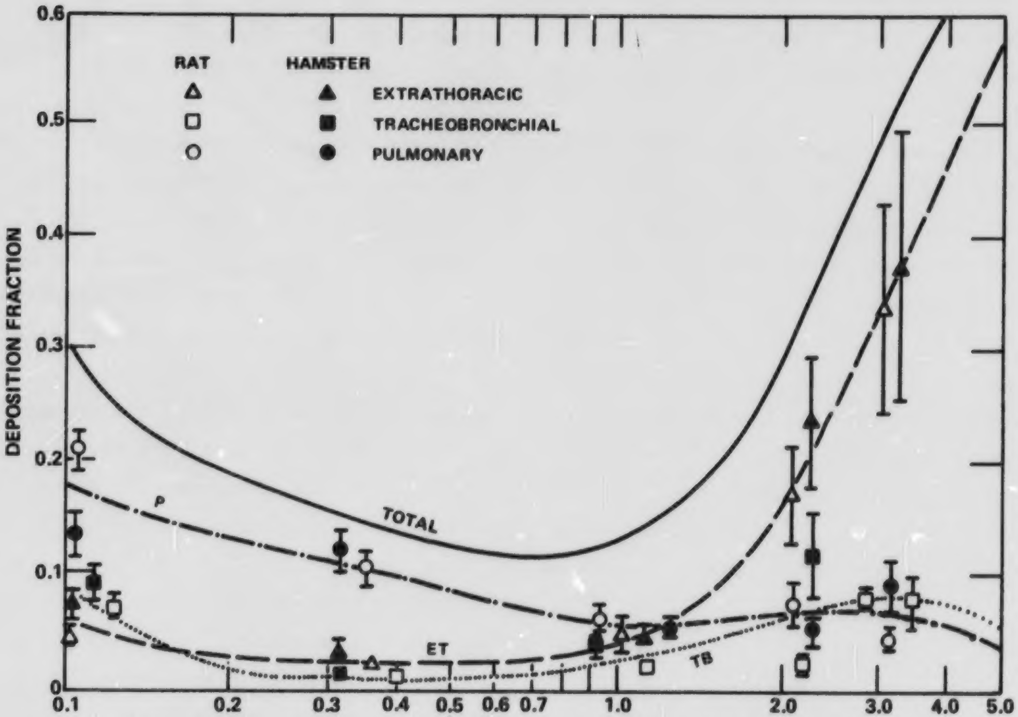


Figure 11-11. Deposition of inhaled monodisperse aerosols of fused aluminosilicate spheres in small rodents showing the deposition in the extrathoracic (ET) region, the tracheobronchial (TB) region, the pulmonary (P) region, and in the total respiratory tract based upon Raabe et al. (1977).

contrast, as presented in Figure 11-7, 50 percent of 5 µm D_{ae} particles inhaled via the mouth deposit in the TB region of humans, showing that large differences can exist between humans and rodents in the TB deposition of large particles. In rodents, the relative distribution among the respiratory regions of particles less than 3 µm D_{ae} during nose breathing follows a pattern that is similar to human regional deposition during nose breathing. Thus, for particles less than 3 µm D_{ae}, the use of rodents or dogs in inhalation toxicology research for extrapolation to humans can be justified from the available data.

11.2.2 Soluble, Deliquescent, and Hygroscopic Particles

Most deposition studies and models tend to focus on insoluble and stable test aerosols whose properties do not change during the course of inhalation and deposition. Environmental aerosols, however, usually contain deliquescent or hygroscopic particles that may grow in the humid respiratory airways. That growth will affect deposition (Scherer et al., 1979). Although the ICRP Task Group on Lung Dynamics (Morrow et al., 1966) addressed this problem by considering the equilibrium diameter for deliquescent materials at relative humidities near, but less than, 100 percent, the residence times in the respiratory tract may be too short for large particles to reach their equilibrium size (Nair and Vohra, 1975; Charlson et al., 1978). Also, environmental aerosols may consist of a combination of components, including complex mixtures, that may not behave like pure substances. Since the temperature of the inspired aerosol will usually be less than that of the respiratory tract environment, supersaturation of water vapor, with respect to the aerosol particles, may exist.

Ferron (1977) described the factors affecting soluble particle growth in the airways during breathing. His results suggest that particles 1 µm D_{ae} will increase by a factor of 3 to 4 in aerodynamic diameter during passage through the airways. Extrathoracic, TB, and P deposition of the enlarged particles would be greater than the deposition expected for the original particle size. Submicrometer particles, including those as small as 0.05 µm, will grow by a factor of 2 in physical diameter, with relatively little effect on deposition. The hygroscopic growth of particles in the diffusion size range (< 0.5 µm physical diameter), however, may alter their deposition pattern substantially, as the diffusional displacement is related to the actual size and not the aerodynamic diameter. Pulmonary deposition of particles smaller than 0.3 µm may be reduced with growth because of reduced diffusivity.

Atmospheric sulfate aerosols can be described as H_2SO_4 partially or completely neutralized by NH_3. Growth of these particles will occur in the respiratory airways during respiration. This growth involves chemical dilution of the electrolyte or acid with absorbed water. A particle growing a factor of 3 in physical diameter must absorb a volume of water equal to 26 times its original particle volume. Also, the increased size will enhance losses by inertial mechanisms, including impaction in the upper airways. A 1 µm D_{ae} particle of H_2SO_4 or $(NH_4)_2SO_4$ may grow to nearly 3 µm D_{ae} in the nasal region, increasing both ET and TB deposition by a factor of 2 or more over the deposition expected for a 1 µm D_{ae} particle, with the net result that P deposition is reduced. Particle growth in the airways may in some cases be protective, since the reduced electrolyte or acid concentration will probably reduce the level of local toxicity.

11.2.3 Surface-coated Particles

Some environmental particles may consist of a relatively insoluble core coated with various chemical species including metallic salts, $(NH_4)_2SO_4$, $(NH_4)HSO_4$, H_2SO_4, organic compounds including polynuclear aromatic hydrocarbons, and small particles of other sparingly soluble materials. Although some surface growth due to water adsorption may occur in the airways, growth will be limited by the availability of deliquescent or hygroscopic components on the particle surface. In general, the increase in aerodynamic diameter that may occur would be much less for coated particles than for purer forms of insoluble materials.

Important examples of coated particles are the fly ash, soot, or other residual solid particulate aerosols released into the environment by combustion of fossil fuels. The exact chemical form of the core of these particles will vary from nearly pure fused aluminosilicate particles produced during the combustion of coal to carbonaceous or metal oxide particles produced by internal combustion engines. Volatile trace metal compounds and organic compounds condense on these particles during the cooling of the effluent stream in the powerplant smokestack or engine exhaust line and during release to the atmosphere. Also, absorption or condensation of SO_2 and other gaseous species from the atmosphere can produce a high surface concentration on particles that are already airborne. If these processes are diffusion-limited, the condensation and coagulation will be quantitatively proportional to particle diameter for particles larger than 0.5 µm D_{ae} and to particle surface area for smaller particles. In either case, the fractional mass of the surface-coating material will be greater on smaller particles than on larger ones. Thus, surface deposition provides a layer of soluble material present in high concentration and results in small-particle enrichment, leading to a shift of the MMD's of the potentially toxic surface materials to smaller aerodynamic equivalent diameters than that of the total particle mass (Natusch and Wallace, 1979). Consequently, pulmonary deposition of surface-enriched material may be significantly enhanced. Important elements such as Se, Cd, As, V, Zn, Sb, and Be have been found to exhibit this size dependence in coal fly ash aerosols (Davison et al., 1974; Natusch et al., 1974; Gladney et al., 1976). (See also Chapters 3, 5, and 6.)

11.2.4 Gas Deposition

The major factors affecting the uptake of gases in the respiratory tract are the morphology of the respiratory tract, the physicochemical properties of the mucous and surfactant layers, the route of breathing and the depth and rate of airflow, physicochemical properties of the gas, and the physical processes that govern gas transport. A brief discussion of these factors serves to illustrate their general role in the deposition of gases and convey some aspects specific to the uptake of SO_2.

The complex morphological structure of the human respiratory tract has been discussed in Section 11.1.3. The nature and structure of the respiratory tract in humans and animals critically influence the deposition of gases, since the relative contribution of gas transport processes varies as a result of this morphology. The human tracheobronchial tree is more symmetric, with respect to diameter ratios and branching angles, than that of dogs, rats, or hamsters but is closest to that of the dog (Phalen et al., 1978). The structure of the tracheobronchial tree is variable from species to species, from lobe to lobe within a given lung, and from one depth to another in the lung.

Physicochemical properties of a gas relevant to respiratory tract deposition are its solubility and diffusivity in mucus, surfactant, lipid, and water, and its reaction-rate constants in mucus, surfactant, lipid, water, and tissue. Henry's law relates the gas phase and liquid phase interfacial concentrations at equilibrium and is a function of temperature and pressure. In general, the more soluble a gas is in biological fluids the higher it is removed in the respiratory tract. Although the solubility of most gases in mucus and surfactant is not known, Henry's law constant for many gases in water is known, the value for SO_2 being 59.7 mole fraction in air per mole fraction in water at $37^{\circ}C$ and one atmosphere of pressure (Washburn, 1928). The diffusivities of most gases in mucus, surfactant, tissue, and water are also unknown, thereby complicating efforts to model gas uptake in the respiratory tract. Diffusivity may be much smaller in a viscous mucous fluid than in water, but ciliary activity induces turbulence, which effectively increases mass transfer. Generally, transport rates of the gas across the mucus-tissue interface, tissue layer, and the tissue-blood interface are needed to fully understand the absorption and desorption of gases in the respiratory tract. Information on biochemical reactions, however, may enable one or more of these compartments to be ignored for a given gas.

The major processes affecting gas transport involve convection, diffusion, and chemical reactions. The bulk movement of inspired gas in the respiratory tract is induced by a pressure gradient and is termed convection. Molecular diffusion due to local concentration gradients is superimposed on this bulk flow at all times, with the transport of the gas being accomplished by the coupling of these two mechanisms. Convection can be decomposed into the processes of advection and eddy dispersion. Advection is the horizontal movement of a mass of air that causes changes in temperature or in other physical properties, and eddy dispersion occurs when air is mixed by turbulence so that individual fluid elements transport the gas and generate the flux. Because of the morphology of the respiratory tract and respiratory airflow patterns, the relative contribution of the various processes to transport and deposition is a function of location and point in the breathing cycle.

During the respiratory cycle, the volumetric flow rate of air varies from zero up to a maximum (dependent upon tidal volume, breathing frequency, and breathing pattern) and then back to zero. Usually expiration is longer than inspiration, and intervening pauses may occur. The net result of these variables is to impart complicated flow patterns and turbulence in some portions of the respiratory tract (see Section 11.1.4).

In the study of the nature of gas mixing in the tracheobronchial tree and its effects on gas transport, numerous modeling efforts have used an approach in which all pathways from the mouth or trachea to the alveoli are combined into one effective pathway whose cross-sectional area is equal to the summed cross-sectional area of all bronchial tubes at a given distance from the mouth or trachea (Davidson and Fitz-Gerald, 1974; Paiva, 1973; Pedley, 1970; Yu, 1975; Scherer et al., 1972). In this formulation, the mechanical mixing imparted by tube bifurcations, turbulence, and secondary flows and the mixing due to molecular diffusion are represented by the functional form of the effective axial diffusion coefficient (Scherer et al., 1975). Thus, this coefficient of diffusion incorporates the effect of axial convection. The effective axial diffusion coefficient is a constant equal to the molecular diffusivity only in the P region, where gas velocity is very small. In other regions of the TB tree, however, the local average gas velocity and the tube geometry will jointly determine the value. Various functional forms were proposed in the studies cited above for an appropriate expression for the effective axial diffusion coefficient.

By constructing individual streamline pathways from the trachea to the alveoli, Yu (1975) derived an expression for the effective axial diffusion coefficient which equalled the algebraic sum of the molecular diffusion coefficient and an apparent diffusion coefficient. The apparent diffusion coefficient arises from two independent mechanisms: 1) the nonhomogeneous ventilation distribution in the lung, and 2) the interaction of nonuniform velocity and concentration profiles due to Taylor's mechanism in individual airways. Using an average standard deviation of airway lengths based on the data of Weibel (1963) and various flow theory limiting values, Yu (1975) demonstrated that Taylor diffusion is dominated everywhere in the TB tree by the apparent diffusion due to nonhomogeneous distribution of ventilation, rather than being a major mechanism for gas transport in some airways as claimed by Wilson and Lin (1970).

In all of the studies described previously, the diffusivity expressions used assume fully developed flow in straight pipes to describe gas mixing, a condition not truly applicable over most of the TB tree. Since flow patterns at tube bifurcations are different for inspiration and expiration (Schroter and Sudlow, 1969), the mixing process and hence the effective diffusivities are different. To obtain diffusivities applicable to the TB tree, Scherer et al. (1975) used airway lengths and diameters from Weibel (1963) and branching angles from Horsfield and Cumming (1967) to construct a five-generation symmetrical branched tube model and to determine experimentally the effective axial diffusivity for laminar flow of a gas as a function of mean axial velocities up to 100 cm/s in the zeroth generation tube. The relationship was approximately linear, and diffusivities for expiration were about one-third those for inspiration. The values obtained by Scherer et al. (1975) for steady flow can be applied to oscillating flow in the TB tree provided the oscillating flow can be considered quasi-steady, i.e., steady at any instant of time. This condition should hold in the first 10 generations whenever flowrates are approximately greater than 0.1 l/s (Jaffrin and Kesic, 1974).

Additional experimental uptake data are needed to obtain a better understanding of the effects of various factors on the transport and removal in the lung of gases such as SO_2. Also needed along with these experimental data are refined theoretical approaches, as well as more flexible computational models, such as that of Pack et al. (1977). The amount of SO_2 removed depends on solubility, the velocity and turbulence of the air, the diffusing capacity across the air-tissue interface and through the tissue, the volume of tissue available for gas storage, and the rate of fluid exchange between these tissues and the storage reservoirs in the body for SO_2 (Aharonson, 1976). The rate-controlling factor in the deposition of SO_2 is probably the vapor pressure of dissolved SO_2 in buffered body fluids.

The diffusion coefficient of SO_2 in air at body temperature is 0.144 cm^2/s at sea level (Fish and Durham, 1971; Sherwood et al., 1975). The complicated flow patterns and turbulence in the upper respiratory tract and upper generations of the TB tree, in combination with high solubility in body fluids, are responsible for the large removal of SO_2 in these regions. Frank et al. (1969) surgically isolated the upper respiratory tract of anesthetized dogs with separate connections for the nose and mouth. Radiolabeled SO_2 (^{35}S) was passed through this isolated ET region for 5 min, and nearly complete removal was observed for concentrations of 2.62 to 131 mg/m^3 (1 to 50 ppm) at a flowrate through the nose of 3.5 l/min. Uptake of the mouth averaged more than 95 percent at 3.5 l/min with SO_2 levels of 2.62 and 26.2 mg/m^3 (1 and 10 ppm). When flow was increased 10-fold to 35 l/min, however, uptake by the mouth fell to under 50 percent. Since the plastic tube through which the gas was delivered was inserted only 2 cm into the mouth, a small fraction of the total pathway through the oral cavity, the use of a tube could not account for the lower uptake observed. Moreover, the results are in agreement with the theory of Aharonson et al. (1974). Strandberg (1964), using a tracheal cannula with two outlets that allowed sampling of inspired and expired air, studied the uptake of SO_2 in the respiratory tract of rabbits. He observed 95-percent absorption in the respiratory tract at 524 mg SO_2/m^3 (200 ppm), but at 0.13 mg SO_2/m^3 (0.05 ppm) absorption was lowered to about 40 percent during inspiration, demonstrating an apparent concentration effect. Absorption of SO_2 at expiration was 98 percent in the 524 mg/m^3 (200 ppm) studies, compared with 80 percent for experiments using 0.13 mg/m^3 (0.05 ppm). Dalhamn and Strandberg (1961) found that rabbits exposed to 262 to 786 mg SO_2/m^3 (100-300 ppm) absorbed 90 to 95 percent of the SO_2. They noted that absorption was to some extent dependent on the technique whereby tracheal air samples were obtained.

Corn et al. (1976) studied the upper respiratory tract deposition of SO_2 in cats and computed mass transfer coefficients that can be used with surface area data to calculate the amount of SO_2 removed in various parts of the respiratory tract. Using a theoretical approach, their own empirical data, and information available from the literature, Aharonson et al. (1974) examined the effect of respiratory airflow rate on nasal removal of soluble vapors. The only assumption made regarding factors affecting local uptake was that there was no back pressure in the blood. Hence, whether the rate of uptake is limited by diffusion through the gas phase, diffusion through the tissue, chemical reactions in the tissue, or local blood flow in the tissues, the analytical approach is valid, as long as the rate of uptake is proportional to the gas phase pressure of the vapor. Their analysis for acetone, ether, ozone, and SO_2 showed that the uptake coefficient, which defines the average flux of soluble vapors into the nasal mucosa per gas-phase unit partial pressure, increases with increasing airflow rate.

In experiments described by Brain (1970b), the amount of SO_2 present in the trachea of dogs increased 32-fold when the airflow rate was increased 10-fold. However, had the uptake coefficient not changed with the flowrate, Aharonson et al. (1974) pointed out that penetration would have increased 500-fold. If the uptake coefficient for SO_2 is concentration dependent, as the data of Strandberg (1964) suggest, increasing airflow rate may increase uptake because higher levels of SO_2 are present along the center of the airstream for the same inspired concentration.

The deposition and clearance of SO_2 has also been studied in _in vitro_ model systems. In a model of the TB airways lined with a simulated airway fluid (bovine serum albumin dissolved in saline), SO_2 was absorbed primarily in the upper third of the simulated airway with only a small fraction of the SO_2 reaching the simulated alveolar or bronchiolar regions (Kawecki, 1978).

Uptake and release of SO_2 in the nose of human subjects breathing 42.2 mg/m^3 (16.1 ppm) through a mask during a 30-minute exposure period was studied by Speizer and Frank (1966). During inspiration, the concentration of SO_2 had dropped 14 percent at a distance 1 to 2 cm within the nose and was too small to detect at the pharynx with the analytical method used. Expired gas in the pharynx was also virtually free of SO_2, but in its transit through the nose the expired air acquired SO_2 from the nasal mucosa. The expired SO_2 concentration at the nose was 5.2 mg/m^3 (2.0 ppm), or about 12 percent of the original mask concentration. In most subjects, the nasal mucosa continued to release small amounts of SO_2 during the first 15 min after the SO_2 exposure ended (see Section 11.3.2).

Melville (1970) exposed humans to SO_2 at levels ranging from 4 to 9 mg/m^3 (1.5 to 3.4 ppm) for periods up to 10 min. Respiratory tract extraction of SO_2 during nose breathing was significantly greater ($p < 0.01$) than during mouth breathing (85 vs. 70 percent, respectively) and was independent of the inspired concentration of SO_2. Andersen et al. (1974) found that at least 99 percent of 65.5 mg SO_2/m^3 (25.0 ppm) was absorbed in the nose of subjects during inspiration. Values obtained after 1 to 3 h of exposure were the same as those obtained after 4 to 6 h of exposure, thereby indicating there was no saturation effect during this period of time.

11.2.5 Aerosol-Gas Mixtures

Gases readily diffuse to the surface of particles and can participate in a variety of surface interactions. Surface adsorption related to temperature and gaseous vapor pressure occurs if adsorption sites for the gas molecules are present on the particles. Such physical adsorption can be described by the Langmuir isotherm or more complex isotherms (Gordieyeff, 1956). In addition, chemical adsorption can occur involving chemical transformations and bonds that enhance transfer of gaseous materials to the particulate phase. Such transformations can include both inorganic and organic vapors. In addition, aerosols of liquid droplets can collect and carry volatile species that are dissolved in the droplets. In these cases, aerosols can serve as vectors carrying molecules of various substances deeper into the airways than would occur if the substances were in their gaseous forms.

Sulfuric acid in the environment may be reduced in acidity by naturally occurring ammonia (NH_3) to form ammonium sulfate $(NH_4)_2SO_4$ and ammonium bisulfate (NH_4HSO_4). Larson et al. (1977) made short-term measurements that suggest that endogenously generated NH_3 gas in the human airways may rapidly and completely neutralize H_2SO_4 aerosols at the concentrations that are normally encountered in the ambient environment. Also, NH_3 is generated from food and excreta in inhalation chambers used to expose experimental animals to H_2SO_4, so that some neutralization of H_2SO_4 in these test atmospheres probably occurs.

Because SO_2 is found in the gas phase of various environmental aerosols, the reactions that occur between SO_2 and aerosols, and the gas-to-particle conversions that may occur, can influence greatly the regional deposition of biologically active chemical species. Since SO_2 is highly soluble in water, droplet aerosols, including those formed by deliquescent particles, will collect dissolved SO_2 and can carry some of the resulting sulfurous acid not neutralized by NH_3 deep into the lung. The presence of certain sulfite species formed by such reactions in environmental aerosols has been suggested (Eatough et al., 1978). Sulfur dioxide is also known to be converted to sulfate by reactions catalyzed by some aerosols, including those containing iron or manganese. The simple adsorption of SO_2 to aerosol surfaces by chemical reaction may lead to the aerosol acting as a vector for transporting SO_2 to the P region.

The deposition of the aerosol and gaseous fractions of the sulfur species can be predicted from the properties of these fractions. Hence, the problem of estimating deposition requires an understanding of the proportion of sulfur species associated with the aerosol fraction and their chemical properties. Since these reactions are dynamic processes, the rate and mechanics of the gas-particle chemical reactions, especially as they may occur in the airways, must be understood in order to predict subsequent biological effects, such as the potentiation of increased airway resistance in guinea pigs with SO_2 by some particles (Amdur and Underhill, 1968).

11.3 TRANSFORMATIONS AND CLEARANCE FROM THE RESPIRATORY TRACT

Particulate material deposited in the respiratory tract may eventually be cleared by the TB mucociliary conveyor or nasal mucous flow to the throat and is either expectorated or swallowed. Other deposited material may be cleared by either the lymphatic system or transfer to the blood. Sulfur dioxide reacts rapidly with biological constituents to produce S-sulfonates (Gunnison and Benton, 1971, see Chapter 12, Section 12.2.1.2.1). The role of clearance as a protective mechanism for the respiratory tract depends on the physicochemical characteristics of the particles (or gaseous species), the site of deposition, and respiratory physiology. If the particles dissolve rapidly in body fluids, their deposition in the nasal turbinates with subsequent absorption into the blood is important, and total deposition of soluble particles may be more critical than regional deposition. For relatively inert and insoluble particles, deposition in the P region, where they may be tenaciously retained, may be more hazardous, unless their action is mediated through ET and TB deposition. The deposition by dissolution of SO_2 in [illegible] ET region may be protective, since it may involve less serious biological effects than deposition in the TB or P airways. Mouth breathing would lessen the ET absorption and increase the SO_2 levels entering the lung. If the particles or SO_2 chemically react with body fluids, transformations of the material can affect clearance. In all respiratory regions, the dissolution of particles competes with other clearance processes.

Since respiratory tract clearance may begin immediately after the initial deposition, the dynamics of retention can become quite complicated when additional deposition is superimposed on clearance phenomena, especially if the deposited material affects clearance mechanisms. Extended or chronic exposures are the general rule for environmental aerosols, and particulate material may accumulate in some portions of the lung (Davies, 1963, 1964a; Walkenhorst, 1967; Einbrodt, 1967).

11.3.1 Deposited Particulate Material

An understanding of regional deposition is requisite to an evaluation of respiratory clearance and a description of the retention of deposited particulate materials. In addition, significant differences may exist between the mechanisms of clearance in different mammalian species. Particle deposition in the ET region is limited primarily to larger particles deposited by inertial impaction. Deposition of various aerosol particles may lead to specific biological effects associated with this region. For particles that do not quickly dissolve or do not react with body fluids, clearance from this region is mechanical. The anterior third of the human nose (where most particles >5 μm may deposit) does not clear except by blowing, wiping, sneezing, or other extrinsic means; and particles may not be removed until 1 or more days after deposition (Proctor and Swift, 1971; Proctor et al., 1969, 1973; Proctor and Wagner, 1965, 1967).

The posterior portions of the human nose, including the nasal turbinates, have mucociliary clearance averaging 4 to 6 mm/min, with considerable variation among individuals (Proctor and Wagner, 1965, 1967; Ewert, 1965; van Ree and van Dishoeck, 1962). Particles are moved with this mucus to the throat and are swallowed or expectorated. Various reactions can occur in the gastrointestinal tract, and some assimilation into the blood is possible even for particles that were relatively insoluble in the nose. The ICRP Task Group (Morrow et al., 1966) adopted a 4-min half-time for physical clearance from the human ET (nasopharyngeal) region by mucociliary transport to the throat and subsequent swallowing.

Soluble particles or droplets are readily assimilated by the mucous membranes of the nose directly into the blood. Solubility is graded from extremely insoluble to instantly soluble, and the dissolution rate constant for an aerosol must be considered for each chemical species.

Since the TB region includes both very large and very small airways, particles of various sizes can be deposited. The retention of deposited materials in this region can differ markedly among individuals and can be affected by such factors as cigarette smoking, pathological abnormalities, or responses to inhaled air pollutants. Clearly, the more rapid the clearance, the less time available for untoward responses or latent injury at the site of original deposition. In mouth breathing of aerosols, such as during smoking or under physical exertion, the beneficial filtering of large particles in the nasal airways is lost, and a greater fraction of these large particles can be deposited in the TB region.

An important characteristic of the TB region is that it is both ciliated and equipped with mucus-secreting cells. Mucociliary clearance mechanisms were reviewed by Schlesinger (1973). For relatively insoluble and inert particles, the primary clearance mechanism for the TB region is mucociliary transport to the glottis, with subsequent swallowing and passage into the gastrointestinal tract. Mucus flow influences the ciliary mucous conveyor (Van As and Webster, 1972; Besarab and Litt, 1970; Dadaian et al., 1971).

The rate of mucus movement is slowest in the finer, more distal airways and greatest in the major bronchi and trachea. The pattern of mucus movement is complex, especially at airway bifurcations, where whirlpools may be found that can slow the clearance of particles (Hilding, 1957). Coughing can accelerate TB clearance by the mucociliary conveyor. The size distribution of particles affects their distribution in the TB tree. The average clearance time for small particles that preferentially deposit deep in the lung is longer than for larger particles, which tend to deposit in the larger airways (Albert et al., 1967, 1973; Camner et al., 1971; Luchsinger et al., 1968). Clearance rates for deposited materials vary considerably among normal healthy adults (Yeates et al. 1975).

The clearance of material in the TB compartment cannot be described by a single rate. Data from experimental studies imply that the larger airways clear with a half-time of about 0.5 h, intermediate airways with a half-time of 2.5 h, and finer airways with a half-time of 5 h (Morrow et al., 1967a; Morrow, 1973). There is also considerable variability among individuals (Camner et al., 1972, 1973a,b; Camner and Philipson, 1972; Albert et al., 1967). Material with slow dissolution rates in the TB compartment will usually not persist longer than about 24 h in healthy humans. Cigarette smoking has been reported under various conditions to either increase, decrease, or have little effect on the efficiency and speed of TB clearance (Camner and Philipson, 1972; LaBelle et al., 1966; Bohning et al., 1975; Albert et al., 1974).

Particles smaller than about 10 μm D_{ae} are deposited to some extent in the P region of the lung on inhalation. Particles that deposit in the P region land on surfaces kept moist by a complex liquid containing surfactants. Slowly dissolving materials that deposit in the human P region are usually retained for years. For example, McEuen and Abraham (1978) reported that birefringent particle counts were significantly higher in 37 cases of pulmonary alveolar proteinosis, both in regions of alveolar proteinosis and in perivascular and peribronchiolar regions (dust retention areas), than in 13 control subjects. Out of 8619 particles, 4817 were < 1 μm in physical diameter, 3771 were 1 to 10 μm in physical diameter, and 31 were > 10 μm in physical diameter, with 59 percent being round, 19 percent fibrous, and 22 percent irregular in shape. Although few particles larger than 10 μm D_{ae} are deposited in the P region, a small number of large aeroallergen particles on the order of 25 μm D_{ae} have been found in the deep lung parenchyma (Michel et al., 1977). Accumulation of pigment in the lungs is reflective of exposure to particulate matter (Pratt and Kilburn, 1971; Sweet et al., 1978).

Usually, relatively insoluble particles are rapidly phagocytized by pulmonary macrophages (LaBelle and Brieger, 1961; Sanders and Adee, 1968; Green, 1971, 1974; Ferin, 1967, 1976, 1977; Camner et al., 1973a,b, 1974; Hibbs, et al., 1977; Ferin et al., 1965; Brain and Corkery, 1977; Brain et al., 1977; Brain, 1970a). Some particles may enter the alveolar interstitium by pinocytosis (Strecker, 1967). Some particles may be cytotoxic to alveolar macrophages and thus influence this clearance mechanism (see Section 12.3.4.2). Migration and grouping of macrophages laden with particles can lead to redistribution of evenly dispersed particles into clumps and focal aggregations of particles in the deep lung. Such events have been described in the sequence of pathological changes observed in experimentally-induced silicosis (Heppleston, 1969). Silica particles ranging in size from less than 1 to 3 μm in physical diameter have been found post mortem in fibrotic lesions associated with deposits of crystalline silica (Craighead and Vallyathan, 1980). Sherwin and coworkers (1979) found an abnormal number of birefringent particles in the lungs of seven patients in association with early to late interstitial inflamation and fibrosis. Also, using scanning electron microscopy and energy dispersive X-ray analysis of particles < 5 μm in physical diameter, they found mostly silicates (especially aluminum, sodium, and potassium), with 5 to 10 percent silicon dioxide.

Macrophages containing particles may enter the boundary region between the ciliated bronchioles and the respiratory ducts and then can be carried with the mucociliary flow of the TB region. Some insoluble particles deposited in the lung are eventually trapped in the P interstitium (Strecker, 1967), impeding mechanical redistribution or removal (Felicetti et al., 1975). Although protein molecules may pass across the air-blood barrier intact with a clearance half-time of hours by pinocytotic vesicular transport (Bensch et al., 1967), there is conflicting evidence at best on the passage of very small particles (< 10 nm in physical diameter) across the air-blood barrier. For example, the data of Kanapilly and Diel (1980) on the dissolution of ultrafine $^{239}PuO_2$ (plutonium dioxide) are in disagreement with the data and interpretations of Raabe et al. (1978).

Another possible clearance route for migrating particles and particle-laden macrophages is the pulmonary lymph drainage system, with translocation first to the TB lymph nodes (Thomas, 1968; Lauweryns and Baert, 1977; Leeds et al., 1971). Little information is available about the clearance rates for transfer from lung to lymph nodes in man. Estimates of half-times range from 1 to 2 y for PuO_2 in dogs and monkeys (Leach et al., 1970) to hours and days for iron, cadmium, and lead in dogs (Oberdörster et al., 1978). The studies by Oberdörster and coworkers (1978) indicate that both the chemical species and their physical states are important in affecting alevolar permeation into the pulmonary lymph. Usually more than 10 percent of the material initially deposited in the P region can be recovered in the regional lymph nodes; and since the amount of material passing through the lymph nodes is not known, nodal values most likely underestimate the role of lymph transport in pulmonary removal (Morrow, 1972). Because the mass of the P lymphoid tissue is only a few percent of the lung weight, the average dose delivered to the lymphoid tissue may be orders of magnitude greater than that delivered to lung tissue. Lymph node retention times usually exceed P retention times, accentuating the disproportionality between lung and lymphoid dose (Morrow, 1972). In addition, Ferin and Feldstein (1978), using inhalation exposures of 15 and 100 mg TiO_2/m^3, showed that the amount of material deposited in the lungs of rats can affect the fraction cleared via the lymphatic system if the exposure level is sufficiently high enough to overwhelm some components of host defenses. As in transfer to the TB region with clearance by the mucociliary escalator, transfer to the lymph nodes may affect only a portion of the material deposited in the lungs. For example, after intratracheal instillation in rats of a mixture of 3, 9, and 15 μm physical diameter (calculated aerodynamic diameters of 3.4., 10.1, and 16.8 μm, respectively) polystyrene latex spheres, Snipes and Clem (1981) found that the 3 μm spheres translocated to the lung-associated lymph nodes, whereas the 9 and 15 μm spheres did not.

Waligora (1971) reported the P clearance of extremely insoluble and inert particles of zirconium oxide radiolabeled with ^{95}Nb. Although his results were not precise, the biological clearance half-life in man was found to be about 1 y, a value about the same as for beagles. By contrast, murine species have a more rapid pulmonary clearance (Morgan et al., 1977). Leach et al. (1970, 1973) exposed experimental animals to insoluble PuO_2 (MMAD of about 3.5 μm) and observed lung retention half-times of 19.9 mo for dogs and 15.5 mo for monkeys. Ramsden et al. (1970) measured the retention of accidentally inhaled, relatively insoluble $^{239}PuO_2$ in a man's lungs and found the clearance half-time to be about 240 to 290 days; some of that material was dissolved into blood and excreted in the urine. Pulmonary clearance half-times as long as 1000 days have been reported for particles of PuO_2 in dogs (Raabe and Goldman, 1979). Cohen et al. (1979) reported an apparent half-time of about 100 days for nonsmokers and about 1 y for smokers for P clearance of magnetite particles. Lung retention studies by Snipes and Clem (1981) of 3, 9, and 15 μm physical diameters polystyrene latex spheres in rats after intratracheal instillation showed that half of the 3 μm spheres were retained with a half-time of 69 days, and 24 and 76 percent of the 9 μm spheres cleared with half-times of 17 and 580 days, respectively. In contrast, 14 percent of the 15 μm spheres cleared with a half-time of 2.3 days and the remaining spheres were retained in the lung with a half-time that was not measurable over the course of the 106-day study. Their results indicate that large particles deposited in the P region could be retained indefinitely and yield unique dose patterns in surrounding tissue.

Because of the slow clearance by the various mechanical pathways, dissolution and associated physical and biochemical transformations are often the dominant mechanisms of clearance from the P region (Morrow, 1973). The term "dissolution" is taken in its broadest context to include whatever processes cause material in a discrete particle to be dispersed into the lung fluids and the blood (Green, 1975). Many chemical compounds deposited in the lung in particulate form are mobilized faster than can be explained by known chemical properties at the normal lung fluid pH of about 7.4 (Kanapilly, 1977). Raabe et al. (1978) suggested that the apparent dissolution of highly insoluble PuO_2 actually may be due to fragmentation into particles small enough to move readily into the blood, rather than to true dissolution.

Mercer (1967) developed an analysis of P clearance based on particle dissolution under nonequilibrium conditions. If the dissolution rate constant (k) is known for a material, the time required to dissolve half the mass of (monodisperse) particles of initial physical diameter (D_o) is given by:

$$T_{1/2} = 0.618\ \alpha_v\ \rho D_o/\alpha_s k \qquad (1)$$

with ρ the physical density of the particles and α_v and α_s the volume and surface shape factors, respectively (for spherical particles $\alpha_s/\alpha_v = 6$).
The particles would be expected to be completely dissolved at a time, t_f, given by:

$$t_f = 3\alpha_v\ \rho\ D_o/\alpha_s k \qquad (2)$$

Mercer (1967) also calculated the expected dissolution half-time for polydisperse particles when their mass median (physical) diameter in the lung is known:

$$T_{1/2} = 0.6\ \alpha_v\ \rho(MMD)/\alpha_s k \qquad (3)$$

Further, he showed that the resulting apparent lung retention function R(t) could be described as the sum of two exponentials of the form:

$$R(t) = f_1 e^{-\lambda_1 \beta} + f_2 e^{-\lambda_2 \beta} \qquad (4)$$

where $f_1 = (1-f_2)$, $\beta = \alpha_s kt/\alpha_v \rho(MMD)$, and f_1, f_2, λ_1, and λ_2 are functions of the geometric standard deviations as defined by Mercer (1967).

For dissolution-controlled P clearance, smaller particles will exhibit proportionately shorter clearance half-times. When the dissolution half-times are much shorter than the half-times associated with the translocations of particles to the TB region or to lymph nodes (i.e., much less than 1 y), dissolution will dominate retention characteristics. Materials usually thought to be relatively insoluble (such as glass) may have high dissolution rate constants and short dissolution half-times for the small particles found in the lung; the dissolution half-time for 1 μm D_{ae} glass spheres is about 75 days (Raabe, 1979). Changes in structure or chemical properties, such as by heat treatment of aerosols (Raabe, 1971), can lead to important changes in dissolution rates and observed P retention.

Usually the retention time of material in the respiratory tract is measured (such as with radiolabeled aerosols) rather than the clearance rates (Sanchis et al., 1972; Camner et al., 1971; Edmunds et al., 1970; Luchsinger et al., 1968; Aldas et al., 1971; Ferin, 1967; Barclay et al., 1938; Morrow et al., 1967a,b; Friberg and Holma, 1961; Holma, 1967; Kaufman and Gamsus, 1974). The lung burden or respiratory tract burden can be represented by an appropriate retention function with time as the independent variable (Morrow, 1970a,b). For models based on simple first-order kinetics, the lung burden, y, at a given time during exposure is controlled by the instantaneous equation:

$$\frac{dy}{dt} = E - \lambda_1 y \qquad (5)$$

where E is the instantaneous deposition rate of particulate material deposited in the lung per unit time during an inhalation exposure and λ_1 is the fraction of material in the lung cleared from the lung per unit time (Raabe, 1967). For an exposure that lasts a time t_e, the lung burden from the exposure is given by:

$$y_e = (E - Ee^{-\lambda_1 t_e})/\lambda_1 \qquad (6)$$

where E in this case, is the average exposure rate. After the exposure ends, the clearance is governed by:

$$\frac{dy}{dt} = - \lambda_1 y \qquad (7)$$

and the lung burden is given by:

$$y = y_e \, e^{-\lambda_1 t} \qquad (8)$$

where y_e is the lung burden at the end of the exposure period (t_e). Hollinger et al. (1979) used this simple model to describe the deposition and clearance of inhaled submicrometer ZnO (zinc oxide) in rats (Figure 11-12) where the concentration of zinc (as Zn) in the lungs (as described by equations 6 and 8) is superimposed on the natural background concentration of zinc in lung tissue. The normally insoluble zinc has only a 4.8-h dissolution half-time

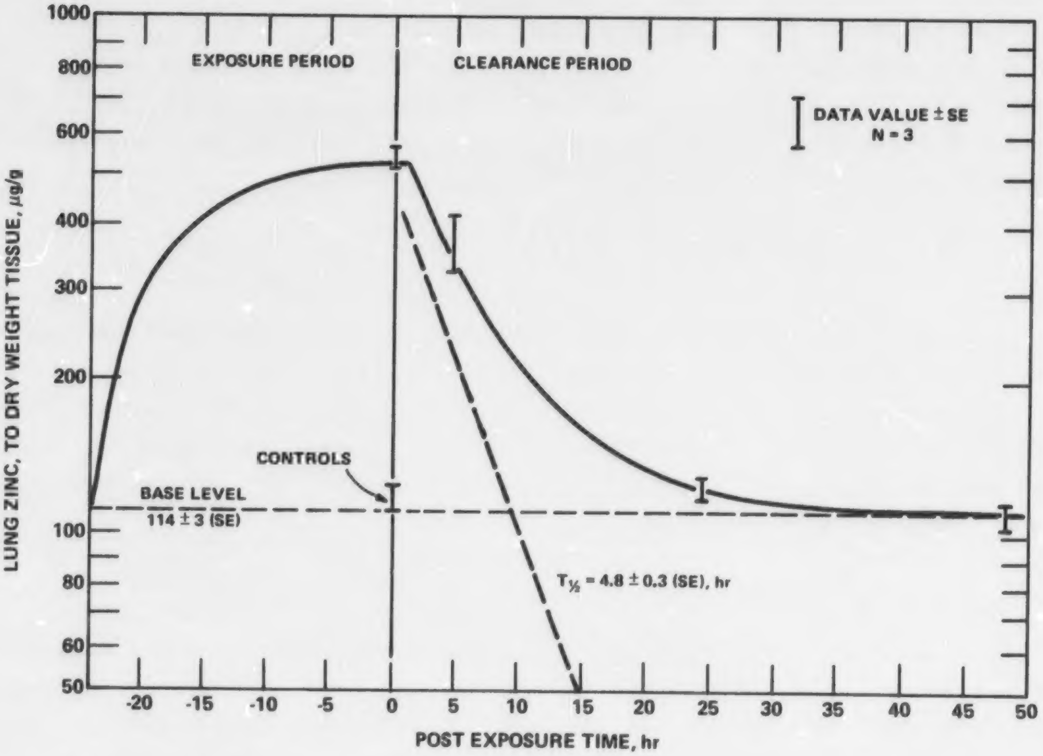


Figure 11-12. Single exponential model, fit by weighted least-squares of the buildup (based on text equation 7) and retention (based on text equation 9) of zinc in rat lungs.

Source: Hollinger et al. (1979).

($\lambda_1 = 0.21\ h^{-1}$) for this aerosol. Of course, environmental aerosol exposures are likely to continue so that a steady state lung burden may be expressed by:

$$y_{ss} = E/\lambda_1 \tag{9}$$

If several deposition and clearance regions, subregions, or special pools are involved, a more complicated multicompartmental model may be required to describe lung or respiratory tract buildup and retention of inhaled aerosols. If each compartment can be described by first order kinetics, a general model can be specified by 1) subscripting E, λ, and Y with the subscript i whenever they appear on the right-hand side of equations 6, 8, and 9, and 2) performing a summation over i from one to the number of compartments. Each of the λ_i values translates to a clearance rate for each of the compartments given by half-time $T_{1/2} = \ln 2/\lambda_i$.

For chronic exposures where the several pools are in complex arrays of change, a simple power function may serve as a satisfactory model of P retention (Downs et al., 1967). In such a model, the P region is treated as one complex, well-mixed pool into which material is added and removed during exposure, as given by the instantaneous equation:

$$\frac{dy}{dt} = E - \lambda_p y/t\ [y = 0 \text{ at } t = 0] \tag{10}$$

where y is the total lung burden at a given time, t, E is the average deposition rate of inhaled particulate material in the lung, and λ_p is the fraction of available lung burden being cleared. Unlike the λ_i of the exponential retention models, λ_p is dimensionless. The time coordinate is not arbitrary; time is taken as zero only at the beginning of the inhalation exposure, when the lung burden is nil. Thus, during an exposure lasting until time (t_e), the P burden (y_e) is given by (Raabe, 1967):

$$y_e = Et_e/(\lambda_p + 1) \tag{11}$$

On this basis, no steady-state concentration is reached even though clearance is progressing and the lung concentration continues to increase during chronic exposures to environmental aerosols. This model is therefore not applicable to relatively soluble species. The lung burden, y, after the exposure has ended for a time, t_p, is given by (Raabe, 1967):

$$y = y_e\ t_e^{\lambda_p}\ t^{-\lambda_p} = At^{-\lambda_p}\ [t = t_e + t_p]. \tag{12}$$

Deposited particulate material cleared from the lung is usually transformed chemically and transferred to other tissues of the body. The injurious properties of a toxic material translocated from the lung may therefore be expressed in other organs. Identification of the potential hazards associated with inhalation exposures to toxicants is compounded when the respiratory tract is not the only target for injury but still serves as the portal of entry into the body. The metabolic behavior and excretion of inhaled toxicants after deposition in the lung may define the probable target organs and indicate potential pathogenesis of resulting disease.

Multicompartmental models that describe biological behavior can become extremely complex. Each toxicant or component of aerosol particles deposited in the respiratory tract may need to be described by a separate rate constant and pool or compartment. A general model of the metabolic behavior of inhaled particles developed by Cuddihy (1969) identified 39 different places where rate constants may need to be determined. In this general model, the P region of the lung is visualized as consisting of three independent clearance compartments, and the particles are presumed to be converted from their original particulate state to some other physicochemical form or transformed state prior to clearance from the respiratory tract. Such a transformed state can be used to describe, for example, the behavior of hydrolyzable aerosols in the respiratory tract.

11.3.2 Absorbed Sulfur Dioxide

Sulfur dioxide coming in contact with the fluids lining the airways (pH 7.4) should dissolve into the aqueous fluid and form some bisulfite (HSO_3^-) and considerable sulfite (SO_3^{2-}) anions. Because of the chemical reactivity of these anions, various reactions are possible, leading to the oxidation of sulfite to sulfate (see Section 12.2.1).

Clearance of sulfite from the respiratory tract may involve several intermediate chemical reactions and transformations (see Section 12.2.1.2). Gunnison and Benton (1971) identified S-sulfonate in blood as a reaction product of inhaled SO_2. The reaction rate is rapid, if not nearly instantaneous, so that there is no long-term clearance to characterize. Intermediate and potentially toxic products may be formed, however. These products may have residence times that are long enough to demonstrate an elevation of the sulfur content of the lung.

Desorption from the upper respiratory tract may be expected whenever the partial pressure of SO_2 on mucosal surfaces exceeds that of the air flowing by. Desorption of SO_2 from mucosal surfaces was still evident after 30 min of flushing with ambient air the airways of dogs that had breathed 2.62 mg/m^3 (1.0 ppm) for 5 min (Frank et al., 1969). Frank et al. (1967) reported SO_2 in the lungs of dogs that apparently was carried by the blood after nasal deposition. In human subjects breathing 42.2 mg/m^3 (16.1 ppm) through a mask for 30 min, 12 percent of the SO_2 taken up by the tissues in inspiration reentered the airstream in expiration and another 3 percent was desorbed during the first 15 min after the end of SO_2 exposure (Speizer and Frank, 1966). Thus, during expiration, SO_2 was desorbed from the nasal mucosa in quantities totaling approximately 15 percent of the amount originally inspired.

The effects of SO_2 on TB clearance in nine healthy, nonsmoking adults were studied by Wolff et al. (1975) (see Section 13.2.3.5). Technetium (Tc) 99 m albumin aerosol (3 µm MMAD, σ_g = 1.6) was inhaled as a bolus under controlled conditions. A 3-h exposure to 13.1 mg

SO_2/m^3 (5.0 ppm) had no significant effect on mucociliary clearance in resting subjects, except for a small transient increase ($p < 0.05$) after 1 hour. A significant decrease in nasal mucus flowrates during a 6-h exposure of 15 young men to 13.1 mg SO_2/m^3 (5.0 ppm) and 65.5 mg SO_2/m^3 (25.0 ppm), but not 2.62 mg SO_2/m^3 (1.0 ppm), was observed by Andersen et al. (1974). Decreases were greatest in the anterior nose and in subjects with initially slow mucus flow rates. Newhouse et al. (1978) assessed the effect of oral exposure to SO_2 on bronchial clearance of a radioactive aerosol (3 μm MMAD) in healthy nonsmoking males and females who exercised periodically during exposure at an exertion level sufficient to keep the heart rate at 70 to 75 percent of the predicted maximum. After a 2-h exposure to 13.1 mg SO_2/m^3 (5.0 ppm), clearance was increased.

11.3.3 Particles and Sulfur Dioxide Mixtures

The presence of adsorbed SO_2 or other sulfur compounds on aerosol surfaces may alter the clearance processes of both. Chemical reactions involving sulfur compounds on particle surfaces may enhance the apparent solubility of the aerosol particles. These aerosol particles may also undergo reaction with sulfite or other species on contact with body fluids.

The formation of sulfate anions by oxidation of SO_2 to SO_3 may be catalyzed by manganese, iron, or other aerosol components. The SO_3 reacts immediately with water to form H_2SO_4 that can react with other materials, such as metal oxides on fly ash aerosols, to produce sulfate compounds. Since sulfate is a normal constituent of body fluids (Kanapilly, 1977), the clearance of sulfate anions probably involves simple dissolution into body fluids.

11.4 AIR SAMPLING FOR HEALTH ASSESSMENT

The objective of air sampling in relation to health assessment is to obtain data on the nature and extent of potential health hazards resulting from the inhalation of airborne particles. To be effective, the techniques used in air samplers must be based on a recognition of the size-selecting characteristics of the human respiratory tract (see Section 11.2). The usual variables affecting the selection of methods, such as the physical limitations of the collection process and the sensitivity and specificity of the analytical procedures, must still be addressed.

An increasing recognition of the importance of the selective sampling of "respirable" dusts has occurred in recent years. The commonly measured index of gross air concentration provides a crude and sometimes misleading indication of health hazard. Since most aerosols are polydisperse, with a $\sigma_g > 2$, the mass median size approaches the diameter of the largest particles in the sample, resulting in a relatively few large particles strongly influencing the value reported for the mass concentration. The measured total mass concentration then will not relate to the inhalation hazard if these particles are not inhaled. Also, the true total airborne mass concentration may be underestimated when the aerosol contains very large particles, since every sampler has its own characteristic upper size cutoff. This cutoff is dependent on its entry shape, dimensions, and flowrate.

The best dose estimates for a substance whose toxicity is proportional to absorbed mass are obtained from information on the mass concentrations within various size ranges. Lippmann (1978) cited several ways such data can be obtained: (1) During the process of collection, separate the aerosol into size fractions that correspond to anticipated regional deposition; (2) analyze the size distribution of the airborne aerosol; and (3) analyze the size distribution of the collected sample. The most reliable and useful information is obtained using methods of fractionation based on aerodynamic diameters similar to the way fractionation occurs within the respiratory tract, thereby automatically compensating for differences in particle shape and density.

Many recent advances have been noted in the technologies needed to develop samplers that will separate particles during the process of collection into "respirable" and "nonrespirable" fractions. The absence of uniform criteria for "respirable" mass concentrations has been a major factor limiting the application of selective sampling concepts in the United States. Regulations established on the basis of only gross concentration limits do not promote field measurements of "respirable" mass.

The recommendations by Miller et al. (1979) on size considerations for establishing a standard for "inhalable" particles indicate a possible future departure from the current approach to the setting of a particulate standard in the United States. Also, the recent report on respirable dust by the International Standards Organization ad hoc Group to TC 146 (1981) contains recommendations for size definitions for particle sampling for the healthy normal segment of the population and high-risk subpopulations. A perspective on these recent events can be obtained by examining the development of the field of respirable dust sampling.

In 1952, the British Medical Research Council (BMRC) adopted a definition of "respirable dust" which essentially considered respirable dust to be that dust reaching the alveoli, thereby making "respirable dusts" applicable to pneumoconiosis-producing dusts. The horizontal elutriator was chosen as a particle size selector, and respirable dust was defined as that dust passing an ideal horizontal elutriator. The elutriator cutoff was chosen to result in the best agreement with experimental lung deposition data. The Johannesburg International Conference on Pneumoconiosis in 1959 adopted the same standard (Orenstein, 1960).

In January 1961, at a meeting in Los Alamos sponsored by the Atomic Energy Commission (AEC) Office of Health and Safety, a second standard was established, which defined "Respirable Dust" as that portion of the inhaled dust which penetrates to the nonciliated portions of the lung (Hatch and Gross, 1964). This definition was not intended to be applicable to dusts that are readily soluble in body fluids or are primarily chemical intoxicants, but rather only for "insoluble" particles that exhibit prolonged retention in the lung. Criteria for respirability were such that all 2 μm D_{ae} particles were considered respirable and particles 10 μm D_{ae} were considered to be nonrespirable.

Other groups, such as the American Conference of Governmental Industrial Hygienists (ACGIH), have incorporated respirable dust sampling concepts in setting acceptable exposure levels for other toxic dusts. Such applications are more complicated, since animal and human exposure data, rather than predictive calculations, form the data base for standards. The size-selector characteristic specified in the ACGIH standard for respirable dust (Threshold Limits Committee, 1968) is almost identical to that of the AEC, differing only at 2 μm D_{ae}, where it allows for 90 percent passing the first-stage collector instead of 100 percent. The difference between them appears to be a recognition of the properties of real particle separators, so that, for practical purposes, the two standards may be considered equivalent (Lippmann, 1978).

The sampler acceptance criteria of the BMRC and of the ACGIH and the P deposition curves from Figure 11-9 are shown in Figure 11-13. The cutoff characteristics of the precollectors preceding respirable dust samplers are defined by these criteria. The two sampler acceptance curves have similar, but not identical, characteristics, due mainly to the use of different types of collectors. The BMRC curve was chosen to give the best fit between the calculated characteristics of an ideal horizontal elutriator and available lung deposition data; on the other hand, the design for the AEC curve was based primarily on the upper respiratory tract deposition data of Brown et al. (1950). The separation characteristics of cyclone type collectors simulate the AEC curve. Whenever the particle size distribution has a $\sigma_g > 2$, samples collected with instruments meeting either criterion will be comparable (Lippmann, 1978). Various comparisons of samples collected on the basis of the two criteria are available (Knight and Lichti, 1970; Breuer, 1971; Maguire and Barker, 1969; Lynch, 1970; Coenen, 1971; Moss and Ettinger, 1970).

The various definitions of respirable dust are somewhat arbitrary, with the BMRC and AEC definitions being based on the "insoluble" particles that reach the P region. Since part of the aerosol that penetrates to the alveoli remains suspended in the exhaled air, respirable dust samples are not intended to be a measure of P deposition but only a measure of aerosol concentration for those particles that are the primary candidates for P deposition. Given that the "respirable" dust standards were intended for "insoluble dusts," most of the samplers developed to satisfy their criteria have been relatively simple two-stage instruments. In addition to an overall size-mass distribution curve, multistage aerosol sampler data can provide estimates of the "respirable" fraction and deposition in other functional regions. Field application of these samplers has been limited because of the increased number and cost of sample analyses and the lack of suitable instrumentation. Many of the various samplers, along with their limitations and deficiencies, were reviewed by Lippmann (1978).

After analyzing industrial health data, Morgan (1978) concluded that different sites of deposition were sufficient to explain the lack of association between pneumoconiosis and bronchitis in coal workers. He found that particles less than 5 μm D_{ae} were associated more with

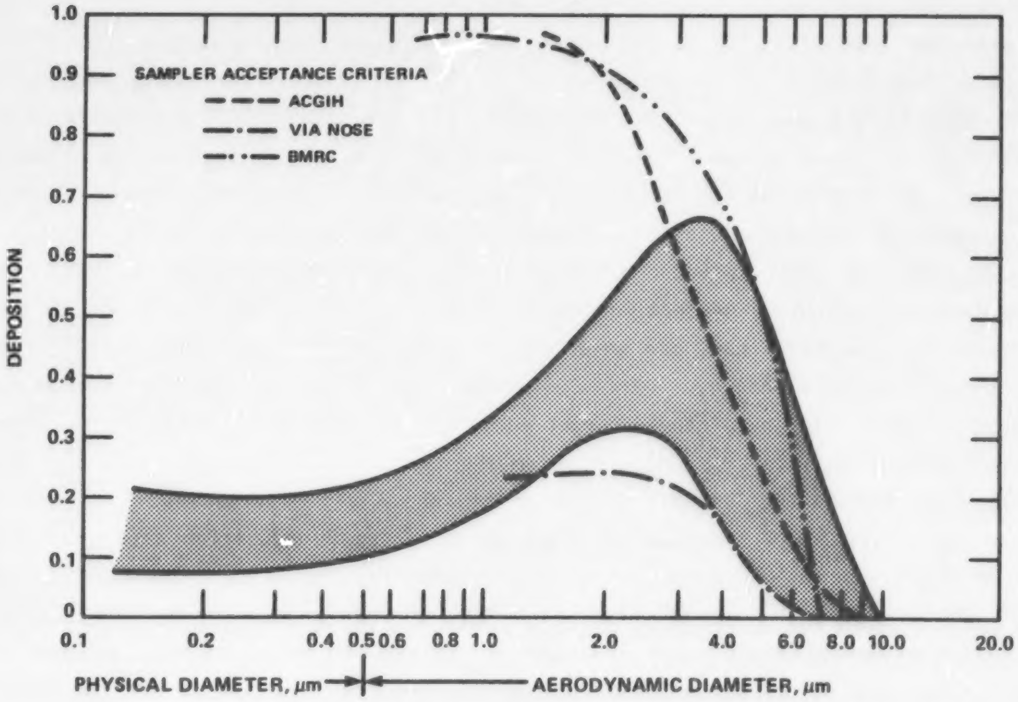


Figure 11-13. Comparison of sampler acceptance curves of BMRC and ACGIH conventions with the band for the experimental pulmonary deposition data of Figure 11-9.

P deposition, as compared to 5 to 10 μm D_{ae} particles, which deposited primarily in areas above the gas-exchange region with nose breathing. In addition, many large particles between 10 and 20 μm D_{ae} were deposited in the trachea and bronchi with mouth breathing. He suggested that when studies are designed for the purpose of relating biological and environmental measurements, it would be wise to measure not only "respirable," but also total dust. Size definitions for particulate sampling which expand the area of concern beyond just "insoluble" dust penetrating to the P region were advanced by Miller et al. (1979) and recently by the International Standards Organization ad hoc Group report to TC 146 (1981). As knowledge of the regional deposition of particles increases through experimental studies, such as those discussed in Section 11.2, it is logical to envisage using samplers that broadly simulate the relative collection efficiencies of the major regions of the respiratory tract. These devices would first select from the total airborne material the inspirable fraction and then sequentially divide this fraction into ET, TB, and P fractions.

Such a division into these three regions (vide supra) was recommended in the International Standards Organization ad hoc Group report to TC 146 (1981). Various options being considered are the at risk population (healthy adults, children, and sick and infirm individuals) and the use of 10 or 15 μm D_{ae} as the 50 percent cut-point for material penetrating to the TB region. Their division of the thoracic fraction into the P and TB fractions, where the target population is healthy adults, is shown in Figure 11-14 using 15 μm D_{ae} as the 50 percent cut-point for the total thoracic fraction. The 50 percent cut-point refers to the aerodynamic diameter for which 50 percent of the particles that enter the mouth or nose are considered to pass the larynx. Thus, the material not passing the larynx forms the ET fraction, which includes the oral pharynx. Particles larger than 15 μm D_{ae} can enter and be deposited in the ET region. If any of these larger particles are readily soluble, they will be absorbed into the bloodstream just as quickly as smaller particles, with one 20 μm D_{ae} particle contributing as much to the systemic dose as a thousand 2 μm D_{ae} particles.

Also shown in Figure 11-14 are bands for the experimental P deposition data of Figure 11-9 and for TB deposition as a percent of particles entering the mouth. The band for TB deposition was derived using the overall regression line of Chan and Lippmann (1980) for ET deposition with mouth breathing and their equation for TB deposition, which was evaluated at bronchial deposition size values one standard deviation from the mean for an average inspiratory flowrate of 30 l/min. The range of values demonstrates the variability among individuals for TB deposition and illustrates the uncertainties as to the particle size corresponding to maximum deposition and the paucity of deposition data for large particle sizes.

The ad hoc group basically followed the BMRC and ACGIH conventions for P deposition in healthy adults, although their definition of the P fraction differs slightly because it is

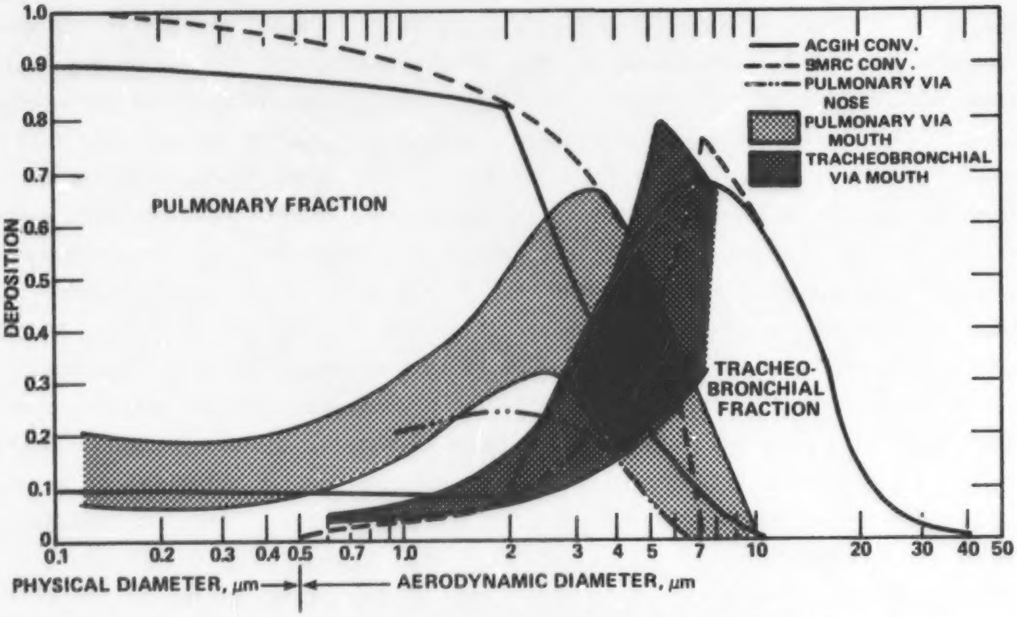


Figure 11-14. Division of the thoracic fraction of deposited particles into pulmonary and tracheobronchial fractions for two sampling conventions (ACGIH and BMRC) as a function of aerodynamic diameter, except below 0.5μm, where physical diameter is used (International Standards Organization, 1981). Also shown are bands for experimental pulmonary deposition data from Figure 11-9 and for tracheobronchial (TB) deposition as a percent of particles entering the mouth. The band for TB deposition was derived using the overall regression line of Chan and Lippmann (1980) for extrathoracic deposition with oral breathing and their equation for TB deposition, which was evaluated at bronchial deposition size values one standard deviation from the mean, given an average inspiratory flow rate of 30 liters per minute. The TB band is shown up to about the largest particle size used by Chan and Lippmann (1980).

Sources: ACGIH (Threshold Limits Committee, 1968); BMRC (Orenstein, 1960); Pulmonary via nose (Lippmann, 1977); Pulmonary via mouth (see Figure 11-9); Tracheobronchial via mouth (Chan and Lippmann, 1980).

defined as a fraction of the inspirable material rather than the total aerosol. A "high-risk" selection curve for children or the sick and infirm used a 50 percent cut-point at 2.5 instead of 3.5 µm D_{ae} in recognition of the fact that a similar shift is seen in lung deposition in these groups (Lippmann, 1977). Taken to its ultimate, by using size selective samplers that separate inspirable material into ET, TB, and P components, standards for airborne particles could specify which of these regional fractions should be measured, taking into account the biological effects of the material, and in the case of the TB and P fractions, the population at risk.

11.5 SUMMARY

Besides being a target of inhaled particles and gases, the respiratory tract is also the portal of entry by which other organs may be affected (see Section 11.3). An understanding of the mechanisms and patterns of translocation to other organ systems is required for evaluation of the potential for injury or response in those organs. When SO_2 or aerosols are inhaled by humans or experimental animals, different fractions of the inhaled materials deposit by a variety of mechanisms in various locations in the respiratory tract. Particle size distribution, particle chemical properties, physicochemical properties of SO_2, respiratory tract anatomy, and airflow patterns all influence the deposition. The three functional regions [extrathoracic (ET) or nasopharyngeal, tracheobronchial (TB), and pulmonary (P)] of the respiratory tract can each be characterized by major mechanisms of deposition and clearance.

Impaction, gravitational settling, and diffusion predominate for the deposition of most types of particles in the respiratory tract, with electrostatic attraction and interception being of relatively minor importance. Diffusivity and interception potential of a particle depend on its geometrical size, whereas the inertial properties of settling and impaction depend on its aerodynamic diameter. Gravitational settling accounts for the deposition of particles in the TB and P regions, and impaction contributes to deposition in the ET and TB regions. Diffusion primarily affects respiratory tract deposition of particles with physical diameters smaller than 1 µm. The major processes affecting the transport of SO_2 in the respiratory tract are convection, diffusion, and chemical reactions. The rapid diffusivity of SO_2 in combination with its high solubility in body fluids is responsible for the large removal of SO_2 in the ET region and upper generations of the TB tree.

After deposition, inhaled particles will be translocated by processes that depend on their character and site of deposition. The anterior third of the human nose does not clear except by blowing, wiping, sneezing, or other extrinsic means, and particles may not be removed until 1 or more days after deposition. If the particles are quite soluble in body fluids, they will readily enter the bloodstream. Relatively insoluble material that lands on ciliated epithelium, either in the ET region or TB airways, will be translocated with mucus flow to the throat and will be swallowed or expectorated. Depending on particle size, relatively insoluble material that deposits on nonciliated surfaces in the P region may be phagocytized, may enter the interstitium and remain in the lung for an extended period, or may be translocated by phagocytic cells, blood, or lymphatic drainage. Some material from the P region may enter the TB region and then be cleared by the mucociliary conveyor. Dissolution can contribute to the clearance of particles in all regions of the respiratory tract.

Nose breathing and mouth breathing provide somewhat contrasting deposition patterns for some respiratory tract regions. With nose breathing, nearly complete respiratory tract deposition can be expected for particles larger than about 4 μm D_{ae}. Since mouth breathing bypasses much of the filtration capabilities of the ET region, there is a shift upward to about 10 μm D_{ae} before there is complete deposition of inhaled particles. Given the three general regions into which the respiratory tract can be divided on the basis of anatomical structure, function, particulate retention times, and clearance pathways, however, regional deposition data for particles of various aerodynamic diameters are more useful than total respiratory tract deposition information.

Particles about 10 μm D_{ae} or larger are deposited in the ET region during nose breathing, as compared with about 65 percent deposition of 10 μm D_{ae} particles under conditions of mouth breathing. On the other hand, for both routes of breathing, ET deposition of particles smaller than about 1 μm D_{ae} is slight. The increased penetration of larger particles deeper into the respiratory tract when a person breathes through the mouth is reflected by experimental deposition data showing that TB deposition of 8 to 10 μm D_{ae} particles is on the order of 20 to 30 percent. Also, about 10 percent of particles as large as 15 μm D_{ae} are predicted to enter the TB region during mouth breathing.

For nose breathing, as compared with mouth breathing, the peak of the P deposition curve shifts downward from 3.5 to about 2.5 μm D_{ae}. Also, the peak is much less pronounced (about 25 percent compared with about 50 percent for mouth breathing), with a nearly constant pulmonary deposition of about 20 percent for all sizes between 0.1 and 4 μm D_{ae}.

The deposition data cited above are based on studies in which healthy young adult subjects usually were used. Although children are usually considered to be a subpopulation more susceptible to the effects of environmental pollutants, deposition data for children are not currently available or likely to be obtained soon. The few data available on other subpopulations, such as asthmatics and chronic bronchitics, indicate that TB deposition appears to be enhanced at the expense of P deposition in most abnormal states. Partial or complete airway obstruction in bronchitis, lung cancer, emphysema, fibrosis, and atelectasis may decrease or eliminate the deposition of particles in some regions of the lungs.

Regional deposition studies of particles less than 3 μm D_{ae} have been conducted using dogs and some rodents. In these species, the relative distribution among the respiratory regions of particles less than 3 μm D_{ae} during nose breathing follows a pattern that is similar to regional deposition in humans during nose breathing. Thus, in this instance, the use of rodents or dogs in toxicological research for extrapolation to humans entails differences in regional deposition of insoluble particles that can be reconciled from available data.

When breathing through the nose under resting conditions, SO_2 removal by nasal absorption is nearly complete in both humans and laboratory animals. Expired air acquires SO_2 from nasal mucosa, with small amounts of SO_2 continuing to be released after cessation of exposure. Extraction of SO_2 by the total respiratory tract during mouth breathing is significantly lower than during nose breathing, although regional uptake has not been studied in humans during mouth or oronasal breathing. On the other hand, studies in which SO_2 was passed through the surgically-isolated ET airways of dogs showed that SO_2 absorption in the ET region can be decreased to less than 50 percent by mouth breathing at elevated airflow rates. Sulfur dioxide may also enter into a variety of gas-to-particle conversions or gas-particle chemical reactions. As a consequence of these reactions with particles, SO_2 can be carried deeper into the respiratory tract, thereby increasing the potential for adverse effects.

Both deposition and retention play roles in determining the effects of inhaled particulate toxicants and SO_2. Everyone is environmentally exposed to a variety of dusts, fumes, sprays, mists, smoke, photochemical particles, and combustion aerosols, as well as SO_2 and other potentially toxic gases. The particle size distribution and chemical and physical composition of airborne particulate material require special attention in toxicological evaluations, since a wide variety of physicochemical properties may be encountered in both experimental and ambient inhalation exposures. The need to characterize the aerosols to which individuals are exposed requires the development of appropriate air sampling techniques so that potential health hazards can be identified. For insoluble dusts whose site of action is the P region, inhalation hazard evaluations based on "respirable" mass are clearly superior to estimates based on gross air concentrations. Appropriate selective sampling procedures can and are being developed to provide more meaningful data on inhalation hazard potential for particles as a function of their regional deposition in the respiratory tract. Gross concentration sampling techniques are appropriate for highly soluble aerosols or where the particle size distribution is relatively constant. They can also be used if the particle size distribution is relatively constant and a known fixed ratio between the gross concentration and the concentration in the size range of interest exists.

11.6 REFERENCES

Aharonson, E. F. Deposition and retention of inhaled gases and vapors. In: Air Pollution and the Lung. E. F. Aharonson, A. Ben-David, and M. A. Klingberg, eds., John Wiley and Sons, New York, NY, 1976. pp. 13-24.

Aharonson, E. F., H. Menkes, G. Gurtner, D. L. Swift, and D. F. Proctor. Effect of respiratory airflow rate on removal of soluble vapors by the nose. J. Appl. Physiol. 27:654, 1974.

Albert, R. E., M. Lippmann,, J. Spiegelman, C. Strehlow, W. Briscoe, P. Wolfson, and N. Nelson. The clearance of radioactive particles from the human lungs. In: Inhaled Particles and Vapours II. C. N. Davies, ed., Pergamon Press, Oxford, England, 1967. p. 361.

Albert, R. E., J. R. Spiegelman, M. Lippmann, and R. Bennett. The characteristics of bronchial clearance in the miniature donkey. Arch. Environ. Health 17:50-58, 1968.

Albert, R. E., J. R. Spiegelman, S. Shatsky, and M. Lippmann. The effect of acute exposure to cigarette smoke on bronchial clearance in the miniature donkey. Arch. Environ. Health 18:30-41, 1969.

Albert, R. E., M. Lippmann, H. T. Peterson, Jr., J. Berger, K. Sanborn, and D. Bohning. Bronchial deposition and clearance of aerosols. Arch. Intern. Med. 131:115-127, 1973.

Albert, R. E., J. Berger, K. Sanborn, and M. Lippmann. Effects of cigarette smoke components on bronchial clearance in the donkey. Arch. Environ. Health 29:96-101, 1974.

Aldas, J. S., M. Dolovich, R. Chalmers, and M. T. Newhouse. Regional aerosol clearance in smokers and nonsmokers. Chest 59:25, 1971.

Altshuler, B. The role of the mixing of intrapulmonary gas flow in the deposition of aerosols. In: Inhaled Particles and Vapours. C. N. Davies, ed., Pergamon Press, Oxford, England, 1961. p. 47.

Altshuler, B., L. Yarmus, E. Palmes, and N. Nelson. Aerosol deposition in the human respiratory tract. AMA Arch. Ind. Health 15:293, 1957.

Altshuler, B., E. D. Palmes, and N. Nelson. Regional aerosol deposition in the human respiratory tract. In: Inhaled Particles and Vapours II. C. N. Davies, ed., Pergamon Press, Oxford, England, 1967. p. 323.

Amdur, M. O., and D. Underhill. The effect of various aerosols on the responses of guinea pigs to sulfur dioxide. Arch. Environ. Health 16:460-468, 1968.

American Heart Association, Committee on Exercise. Exercise testing: performance and interpretation. Ind. Med. Surg. 42:20-28, 1973.

Andersen, I., G. R. Lundquist, P. L. Jensen, and D. F. Proctor. Human response to controlled levels of sulfur dioxide. Arch. Environ. Health 28:31-39, 1974.

Bake, B., L. Wood, B. Murphy, P. Macklem, and J. Milic-Emili. Effect of inspiratory flow rate on regional distribution of inspired gas. J. Appl. Physiol. 37:8, 1974.

Balis, J. V., S. A. Shelley, M. J. McCue, and E. S. Rappaport. Mechanisms of damage to the lung surfactant system. Exp. Molec. Pathol. 14:243, 1971.

Barclay, A. E., K. J. Franklin, and R. G. Macbeth. Roentgenographic studies of the excretion of dusts from the lungs. Am. J. Roentgenol. Radium Ther. 39:673, 1938.

Bartlett, D., J. E. Remmers, and H. Gautier. Laryngeal regulation of respiratory airflow. Resp. Physiol. 18:194, 1973.

Batchelor, G. K. Symmetrical contraction on isotropic turbulence. In: The Theory of Homogeneous Turbulence. Cambridge University Press, London, England, 1953. p. 74.

Beeckmans, J. B. The deposition of aerosols in the respiratory tract. Can. J. Physiol. Pharmacol. 43:157, 1965.

Bell, K. A. Local particle deposition in respiratory airway models. In: Recent Development in Aerosol Science. D. T. Shaw, ed., John Wiley and Sons, New York, NY, 1978. pp. 97-134.

Bell, K., and S. Friedlander. Aerosol deposition in models of a human lung bifurcation. Staub-Reinhalt. Luft 33:183, 1973.

Bensch, K. G., E. Dominguez, and A. A. Liebow. Absorption of intact protein molecules across the pulmonary air-tissue barrier. Science 157:1204-1206, 1967.

Besarab, A., and M. Litt. Model studies on the adhesive properties of mucus and similar polymer solution. Arch. Intern. Med. 126:504, 1970.

Blank, M., A. B. Goldstein, and B. B. Lee. The surface properties of lung extract. J. Colloid. Interface Sci. 29:148, 1969.

Bohning, D. E., R. E. Albert, M. Lippmann, and W. M. Foster. Tracheobronchial particle deposition and clearance. A study of the effects of cigarette smoking in monozygotic twins. Arch. Environ. Health 30:457, 1975.

Brain, J. D. Free cells in the lungs--some aspects of their role, quantitation and regulatory. Arch. Intern. Med. 126:477-487, 1970a.

Brain, J. D. The uptake of inhaled gases by the nose. Ann. Otol. Rhinol. Laryngol. 79:529-539, 1970b.

Brain, J. D., and G. C. Corkery. The effect of increased particles on the endocytosis of radiocolloids by pulmonary macrophages in vivo: competitive and toxic effects. In: Inhaled Particles IV, Part 2. W. H. Walton, ed., Pergamon Press, Oxford, England, 1977. pp. 551-564.

Brain, J. D., J. J. Godleski, and S. P. Sorokin. Quantification, origin and fate of pulmonary macrophages. In: Respiratory Defense Mechansims. J. D. Brain, D. F. Proctor and L. M. Reid, eds., Marcel Dekker, New York, NY, 1977. pp. 849-892.

Breuer, H. Problems of gravimetric dust sampling. In: Inhaled Particles III. W. H. Walton, ed., Unwin Bros., London, England, 1971. pp. 1031-1042.

Brown, J. H., K. M. Cook, F. G. Nex, and T. Hatch. Influence of particle size upon the retention of particulate matter in the human lung. Am. J. Public Health 40:450, 1950.

Camner, P., and K. Philipson. Tracheobronchial clearance in smoking-discordant twins. Arch. Environ. Health 25:60-63, 1972.

Camner, P. and B. Bakke. Nose or mouth breathing? Environ. Res. 21:394-398, 1980.

Camner, P., K. Philipson, L. Friberg, and B. Holma. Human tracheobronchial clearance studies. Arch. Environ. Health 22:444, 1971.

Camner, P., K. Philipson, and L. Friberg. Tracheobronchial clearance in twins. Arch. Environ. Health 24:82, 1972.

Camner, P., P. Hellstrom, and M. Lundborg. Coating 5 mm particles with carbon and metals for lung clearance studies. Arch. Environ. Health 27:331, 1973a.

Camner, P., P. Hellstrom, and K. Philipson. Carbon dust and mucociliary clearance. Arch. Environ. Health 26:294, 1973b.

Camner, P., M. Lundborg, and P. Hellstrom. Alveolar macrophages and 5 mm particles coated with different metals. Arch. Environ. Health 29:211, 1974.

Chan, T. L., and M. Lippmann. Experimental measurements and empirical modelling of the regional deposition of inhaled particles in humans. Am. Ind. Hyg. Assoc. J. 41:399-409, 1980.

Chan, T. L., M. Lippmann, V. R. Cohen, and R. B. Schlesinger. Effect of electrostatic charge on particle deposition in a hollow cast of the human larynx-tracheobronchial tree. J. Aerosol Sci. 9:463, 1978.

Charlson, R. J., D. S. Covert, T. V. Larson, and A. P. Waggoner. Chemical properties of tropospheric sulfur aerosols. Atmos. Environ. 12:39, 1978.

Cheng, Y. S., and C. S. Wang. Inertial deposition of particles in a bend. J. Aerosol Sci. 6:139, 1975.

Cinkotai, F. F. Fluid flow in a model alveolar sac. J. Appl. Physiol. 37:249, 1974.

Clement, J., M. Afschrift, J. Pardens, and K. Van de Woestline. Peak expiratory flow rate and rate of change of pleural pressure. Resp. Physiol. 18:222, 1973.

Clements, J. A., and D. F. Tierney. Alveolar instability associated with altered surface tension. In: Handbook of Physiology. W. D. Fenn and H. Rahn, eds., Section 3, Respiration, vol. II, chapter 69 (American Physiological Society), Washington, DC, 1965. pp. 15-84.

Coenen, W. Berechnung von Umrechnungsfaktorem für verscheidene Feinstaubmessverfahren. [Estimation of conversion factors for different fine dust measurement methods.] In: Inhaled Particles III. W. H. Walton, ed., Unwin Bros., London, England, 1971. pp. 1045-1050.

Cohen, D., S. F. Arai, and J. D. Brain. Smoking impairs long-term dust clearance from the lung. Science, 204:514-516, 1979.

Cohen, V. R. The effects of glyceryl guaiacolate on bronchial clearance in patients with chronic bronchitis. M.S. Thesis, New York University, New York, NY, 1977.

Corn, M., N. Kotsko, and D. Stanton. Mass-transfer coefficient for sulphur dioxide and nitrogen dioxide removal in cat upper respiratory tract. Ann. Occup. Hyg. 19:1, 1976.

Craighead, F. E., and N. V. Vallyathan. Cryptic pulmonary lesions in workers occupationally exposed to dust containing silica. J. Am. Med. Assoc. 244: 1939-1941, 1980.

Cuddihy, R. G. Analog simulation of the biological behavior of inhaled radionuclides. In: Fission Product Inhalation Program Annual Report 1968-1969. LF-41, Lovelace Foundation, Albuquerque, NM, 1969. p. 136.

Cuddihy, R. G., D. C. Brownstein, O. G. Raabe, and G. M. Kanapilly. Respiratory tract deposition of inhaled polydisperse aerosols in beagle dogs. J. Aerosol Sci. 4:35, 1973.

Dadaian, J. H., S. Yin., and G. A. Laurenzi. Studies of mucus flow in the mammalian respiratory tract. Am. Rev. Resp. Dis. 103:808, 1971.

Dalhamn, T., and L. Strandberg. Acute effect of sulfur dioxide on rate of ciliary beat in trachea of rabbit *in vivo* and *in vitro*, with studies on absorptional capacity of nasal cavity. Int. J. Air Water Pollut. 4:154, 1961.

Dautrebande, L., and W. Walkenhurst. New studies on aerosols XXIV. Arch. Int. Pharmacodyn. 162:194, 1966.

Davidson, M. R., and J. M. Fitz-Gerald. Transport of O_2 along a model pathway through the respiratory region of the lung. Bull. Math. Biol. 36:275-303, 1974.

Davies, C. N. A formalized anatomy of the human respiratory tract. *In*: Inhaled Particles and Vapours. C. N. Davies, ed., Pergamon Press, Oxford, England, 1961. p. 82.

Davies, C. N. The handling of particles by the human lungs. Br. Med. Bull. 19:49, 1963.

Davies, C. N. A comparison between inhaled dust and the dust recovered from human lungs. Health Phys. 10:1029, 1964a.

Davies, C. N. Deposition and retention of dust in the human respiratory tract. Ann. Occup. Hyg. 7:169, 1964b.

Davies, C. N. An algebraical model for the deposition of aerosols in the human respiratory tract during steady breathing. J. Aerosol Sci. 3:297, 1972.

Davies, C. N., J. Heyder, and M. C. Subba Ramu. Breathing of half-micron-aerosols. I. Exp. J. Appl. Physiol. 32:592-600, 1972.

Davison, R. L., D. F. S. Natusch, J. R. Wallace, and C. A. Evans, Jr. Trace elements in fly ash: Dependence of concentration on particle size. Environ. Sci. Technol. 8:1107-1113, 1974.

Deal, E. C., E. R. McFadden, R. H. Ingram, R. H. Strauss, and J. J. Jaeger. Role of respiratory heat exchange in production of exercise-induced asthma. J. Appl. Physiol: Resp. Environ. Exercise Physiol. 46:467, 1979a.

Deal, E. C., E. R. McFadden, R. H. Ingram, and J. J. Jaeger. Hyperpnea and heat flux: initial reaction sequence in exercise-induced asthma. J. Appl. Physiol: Resp. Environ. Exercise Physiol. 46:476, 1979b.

Deal, E. C., E. R. McFadden, R. H. Ingram, and J. J. Jaeger. Esophageal temperature during exercise in asthmatic and nonasthmatic subjects. J. Appl. Physiol: Resp. Environ. Exercise Physiol. 46:484, 1979c.

Dekker, E. Transition between laminar and turbulent flow in human trachea. J. Appl. Physiol. 16:1060, 1961.

Doershuk, C. F., T. D. Downs, L. W. Matthews, and M. D. Lough. A method for ventilatory measurements in subjects 1 month to 5 years of age: normal results and observations in disease. Pediatr. Res. 4:165-174, 1970.

Doershuk, C. F., B. J. Fisher, and L. W. Matthews. Pulmonary physiology of the young child. *In*: Pulmonary Physiology of the Fetus, Newborn and Child. E. M. Scarpelli, ed., Mea and Febiger, Philadelphia, PA, 1975.

Downs, W. L., H. B. Wilson, G. Z. Sylvester, L. J. Leach, and E. A. Maynard. Excretion of uranium by rats following inhalation of uranium dioxide. Health Phys. 13:445, 1967.

DuBois, A. B., and R. M. Rogers. Respiratory factors determining the tissue concentrations of inhaled toxic substances. Resp. Physiol. 5:34, 1968.

Dunnill, M. S. Postnatal growth of the lung. Thorax 17:329, 1962.

Eatough, D. J., T. Major, J. Ryder, M. Hill, N. F. Mangelson, N. L. Eatough, L. D. Hansen, R. G. Meisenheimer, and J. W. Fischer. The formation and stability of sulfite species in aerosols. Atmos. Environ. 12:263-271, 1978.

Edmunds, L. H., P. D. Graf, S. S. Sagel, and R. H. Greenspan. Radiographic observations of clearance of tantalum and barium sulfate particles from airways. Invest. Radiol. 5:131, 1970.

Einbrodt, H. J. Experiments on the elimination of dust from human lungs. Ann. Occup. Hyg. 10:47, 1967.

Engel, L. A., L. D. Wood, G. Utz, and P. T. Macklem. Gas mixing during inspiration. J. Appl. Physiol. 35:18, 1973.

Ewert, G. On the mucus flow rates in the human nose. Acta Otolaryngol; Suppl. 200, 1965.

Felicetti, S. A., S. A. Silbaugh, B. A. Muggenberg, and F. F. Hahn. Effect of time post-exposure on the effectiveness of bronchopulmonary lavage in removing inhaled ^{114}Ce in fused clay from beagle dogs. Health Phys. 29:89, 1975.

Ferin, J. The mechanism of elimination of deposited particles from the lungs. Ann. Occup. Hyg. 10:207, 1967.

Ferin, J. Lung clearance of particles. In: Air Pollution and The Lung. E. F. Aharonson, A. Ben-David, and M. A. Klingberg, eds., Halsted Press-John Wiley, Jerusalem, Israel, 1976. pp. 64-78.

Ferin, J. Effect of particle content of lung on clearance pathways. In: Pulmonary Macrophages and Epithelial Cells. Proceedings of the Sixteenth Annual Hanford Biology Symposium, Energy Research and Development Administration and Battelle Memorial Institute, Richland, Washington, September 27-29, 1976. C. L. Sanders, R. P. Dagle, and H. A. Ragan, eds., ERDA Symposium Series 43, Energy Research and Development Administration, Oak Ridge, TN, 1977. pp. 414-428.

Ferin, J. and M. L. Feldstein. Pulmonary clearance and hilar lymph node content in rats after particle exposure. Environ. Res. 16:342-352, 1978.

Ferin, J., G. Urbankova, and A. Vickova. Pulmonary clearance and the clearance of macrophages. Arch. Environ. Health 10:790, 1965.

Ferron, G. A. The size of soluble aerosol particles as a function of the humidity of the air. Application to the human respiratory tract. J. Aerosol Sci. 8:251-267, 1977.

Findeisen, W. Uber das absetzen kleiner in der Luft suspendieter Teilchen in der menschlichen Lunger bei der-Atmung. [On the settling of squall suspended in air particles deposited in the human lung by respiration.] Pflugers Arch. J. d. Physiol. 236:367, 1935.

Fish, B. R., and J. L. Durham. Diffusion coefficient of SO_2 in air. Environ. Lett. 2:13-21, 1971.

Foord, N., A. Black, and M. Walsh. Regional deposition of 2.5 - 7.5 mm diameter inhaled particles in healthy male non-smokers. AERE Harwell, ML. 76:2892, 1976.

Fowler, J. F., and A. E. Young. The average density of healthy lung. Am. J. Roentgenol. Radium Ther. 81:312, 1959.

Frank, N. R., R. E. Yoder, E. Yokoyama, and F. E. Speizer. The diffusion of $^{35}SO_2$ from tissue fluids into the lungs following exposure of dogs to $^{35}SO_2$. Health Phys. 13:31-36, 1967.

Frank, N. R., R. E. Yoder, J. D. Brain, and E. Yokoyama. SO_2 (^{35}S labeled) absorption by the nose and mouth under conditions of varying concentration and flow. Arch. Environ. Health 18:315-322, 1969.

Fraser, D. A. The deposition of unipolar charged particles in the lungs of animals. Arch. Environ. Health 13:152, 1966.

Fraser, R. G., and J. A. P. Pare. Structure and Function of the Lung. W. B. Saunders Co., Philadelphia, PA, 1971.

Friberg. L., and B. Holma. External measurement of lung clearance. Arch. Environ. Health 3:56, 1961.

Fry, D. A preliminary model for simulating the aerodynamics of the bronchial tree. Comp. Biomed. Res. 2:111, 1968.

Fry, F. A., and A. Black. Regional deposition and clearance of particles in the human nose. J. Aerosol Sci. 4:113, 1973.

Fuchs, N. A. The Mechanics of Aerosols. The Macmillan Company, New York, NY, 1964.

George, A. C., and A. J. Breslin. Deposition of natural radon daughters in human subjects. Health Phys. 13:375, 1967.

Giacomelli-Maltoni, G., C. Melandri, V. Prodi, and G. Tarroni. Deposition efficiency of monodisperse particles in human respiratory tract. Am. Ind. Hyg. Assoc. J. 33:603, 1972.

Gil, J., H. Bachofen, P. Gehr, and E. R. Weibel. Alveolar volume-surface area relation in air- and saline-filled lungs fixed by vascular perfusion. J. Appl. Physiol.: Resp. Environ. Excercise Physiol. 47:990-1001, 1979.

Gladney, E. S., J. A. Small, G. E. Gordon, and W. H. Zoller. Composition and size distribution of in-stack particulate material at a coal-fired power plant. Atmos. Environ. 10:1071, 1976.

Goldberg, I. S., and R. V. Lourenco. Deposition of aerosols in pulmonary disease. Arch. Intern. Med. 131:88-91, 1973.

Goldberg, I. S., K. Y. Lam, B. Bernstein, and H. O. Hutchens. Solution to the Fokker-Planck equations governing simultaneous diffusion and gravitational settling of aerosol particles from stationary gas in a horizontal tube. J. Aerosol Sci. 9:209, 1978.

Gordieyeff, V. A. The adsorption of gases and vapors on aerosol particulates. Am. Ind. Hyg. Assoc. Q. 17:411, 1956.

Gormley, P. G., and M. Kennedy. Diffusion from a stream flowing through a cylindrical tube. Proc. R. Ir. Acad. A52:163, 1949.

Grant, B. J., H. A. Jones, and J. M. Hughes. Sequence of regional filling during a tidal breath in man. J. Appl. Physiol. 37:158-65, 1974.

Green, G. M. Alveolobronchiolar transport observations and hypothesis of a pathway. Chest 59:15, 1971.

Green, G. M. In defense of the lung. Am. Lung Assoc. Bull. 60:4, 1974.

Green, J. F. The pulmonary circulation. In: The Peripheral Circulations. R. Zelis, ed., Grune and Stratton, New York, NY, 1975. p. 9.

Gunnison, A. F., and A. W. Benton. Sulfur dioxide: sulfite. Interaction with mammalian serum and plasma. Arch. Environ. Health 22:381-388, 1971.

Guyton, A. C. Measurement of the respiratory volumes of laboratory animals. Am. J. Physiol. 150:20, 1947a.

Guyton, A. C. Analysis of respiratory patterns in laboratory animals. Am. J. Physiol. 150:78, 1947b.

Harris, R. L., and D. A. Fraser. A model for deposition of fibers in the human respiratory system. Am. Ind. Hyg. Assoc. J. 37:73, 1976.

Hatch, T. E., and P. Gross. Pulmonary Deposition and Retention of Inhaled Aerosols. Academic Press, New York, NY, 1964.

Henderson, R. F., J. J. Waide, and R. C. Pfleger. Replacement time for alveolar lipid removed by pulmonary lavage: effects of multiple lavage on lung lipids. Arch. Int. Physiol. Biochim. 83:261, 1975.

Heppleston, A. G. The fibrogenic action of silica. Br. Med. Bull. 25:282, 1969.

Heyder, J. Conditions for the determination of aerosol particle deposition in the human respiratory tract. Staub-Rein. Luft 31:11, 1971.

Heyder, J., and C. N. Davies. The breathing of half micron aerosols - III dispersion of particles in the respiratory tract. J. Aerosol Sci. 2:437, 1971.

Heyder, J., and J. Gebhart. Gravitational deposition of particles from laminar aerosol flow through inclined circular tubes. J. Aerosol Sci. 8:289, 1977.

Heyder, J. L., G. J. Armbruster, and W. Stahlhofen. Deposition of aerosol particles in the human respiratory tract. In: Aerosol in Physik, Medizin and Technik, Proceedings of a Conference, Bad Soden, October 17-18, 1973. V. Bohlan, ed., Gesellschaft fuer Aerosolforschung, Bad Soden, West Germany, 1973a. pp. 122-125.

Heyder, J., J. Gebhart, G. Heigiver, C. Roth and W. Stahlhofen. Experimental studies of the total deposition of aerosol particles in the human respiratory tract. J. Aerosol. Sci. 4:191-208, 1973b.

Heyder, J., L. Armbruster, J. Gebhart, E. Grein, and W. Stahlhofen. Total deposition of aerosol particles in the human respiratory tract for nose and mouth breathing. J. Aerosol Sci. 6:311, 1975.

Heyder, J., J. Gebhart, C. Roth, W. Stahlhofen, B. Stuck, G. Tarroni, T. DeZaiacomo, M. Formignani, C. Melandri, and V. Prodi. Intercomparison of lung deposition data for aerosol particles. J. Aerosol. Sci. 9:147-155, 1978.

Heyder, J., J. Gebhart, and W. Stahlhofen. Inhalation of aerosols. Particle deposition and retention. In: Generation of Aerosols and Facilities for Exposure Experiments. K. Willeke, ed., Ann Arbor Science, Ann Arbor, MI, 1980a. pp. 65-103.

Heyder, J., J. Gebhart, G. Rudolf and W. Stahlhofen. Physical factors determining particle deposition in the human respiratory tract. J. Aerosol Sci., 11:505-515, 1980b.

Hibbs, J. B., R. R. Taintor, H. A. Chapman, and J. B. Weinberg. Macrophage tumor killing: influence of the local environment. Science 197:279-282, 1977.

Higgs, B. E., M. Clode, G. J. R. McHardy, N. L. Jones, and E. J. M. Campbell. Changes in ventilation gas exchange and circulation during exercise in normal subjects. Clin. Sci. 32:329-337, 1967.

Hilding, A. C. Ciliary streaming in the bronchial tree and the time element in carcinogenesis. N. Engl. J. Med. 256:634-640, 1957.

Hollinger, M. A., O. G. Raabe, S. N. Giri, M. Freywald, S. V. Teague, and B. Tarkington. Effect of the inhalation of zinc and dietary zinc on paraquat toxicity in the rat. Toxicol. Appl. Pharmacol. 49:53-61, 1979.

Holma, B. Lung clearance of mono- and di-disperse aerosols determined by profile scanning and whole-body counting. Acta Medica Scand. Suppl. 473, 1967.

Holmes, T. H., H. Goodell, S. Wolf, and H. G. Wolff. The Nose. Charles C. Thomas, Springfield, Ill., 1950.

Horsfield, K., and G. Cumming. Angles of branching and diameters of branches in the human bronchial tree. Bull. Math. Biophys. 29:245, 1967.

Horsfield, K., and G. Cumming. Morphology of the bronchial tree in man. J. Appl. Physiol. 24:373, 1968.

Horsfield, K., G. Dart, D. E. Olson, G. F. Filley, and G. Cumming. Models of the human bronchial tree. J. Appl. Physiol. 31:207-217, 1971.

Hounam, R. F. The deposition of atmospheric condensation nuclei in the nasopharyngeal region of the human respiratory tract. Health Phys. 20:219, 1971.

Hounam, R. F., A. Black, and M. Walsh. Deposition of aerosol particles in the nasopharyngeal region of the human respiratory tract. Nature 221:1254-1255, 1969.

Hounam, R. F., A. Black, and M. Walsh. The deposition of aerosol particles in the nasopharyngeal region of the human respiratory tract. J. Aerosol Sci. 2:47, 1971a.

Hounam, R. F., A. Black, and M. Walsh. The deposition of aerosol particles in the nasopharyngeal region of the human respiratory tract. In: Inhaled Particles III. W. H. Walton, ed., Unwin Brothers Limited, Surrey, England, 1971b. p. 71.

Hughes, J. M., F. G. Hoppin, and J. Mead. Effect of lung inflation on bronchial length and diameter in excised lungs. J. Appl. Physiol. 32:25, 1972.

Intermountain Thoracic Society. Clinical Pulmonary Function Testing: A Manuel on Uniform Laboratory Procedures for the Intermountain Area. R. E. Kanner and A. H. Morris, eds., Salt Lake City, UT, 1975. pp. VI-1 to 140.

International Commission on Radiological Protection. Report of the Task Group on Reference Man. Pergamon Press, Oxford, England, 1975.

International Standards Organization. Size definitions for particulate sampling: recommendations of ad hoc working group appointed by TC 146 of the International Standards Organiation. Am. Ind. Hyg. Assoc. J. 42:A-64 to A-68, 1981.

Jaffrin, M. Y., and P. Kesic. Airway resistance: a fluid mechanical approach. J. Appl. Physiol. 36:354-361, 1974.

Johnston, J. R., and D. C. F. Muir. Inertial deposition of particles in the lung. J. Aerosol Sci. 4:269, 1973.

Johnston, J. R., and R. C. Schroter. Deposition of particles in model airways. J. Appl. Physiol.: Resp. Environ. Exercise Physiol. 47:947-953, 1979.

Jones, N. L., E. J. M. Campbell, R. H. T. Edwards, and D. G. Robertson. Clinical Exercise Testing. W. B. Saunders, Philadelphia, PA, 1975. p. 202.

Kanapilly, G. M. Alveolar microenvironment and its relationship to the retention and transport into blood of aerosols deposited in the alveoli. Health Phys. 32:89, 1977.

Kanapilly, G. M., and J. H. Diel. Ultrafine $^{239}PuO_2$ aerosol generation, characterization and short-term inhalation study in the rat. Health Phys. 39:505-519, 1980.

Kaufman, L., and G. Gamsus. Fluorescent excitation in the measurement of clearance of heavy metals from the lungs. I.E.E.E. Trans. on Nuclear Sci. NS-21:1721, 1974.

Kawecki, J. M. Emission of Sulfur-Bearing Compounds from Motor Vehicle and Aircraft Engines, A Report to Congress. EPA-600/9-78-028, U.S. Environmental Protection Agency, Research Triangle Park, NC, August 1978.

Klass, D. J. Immunochemical studies of the protein fraction of pulmonary surface active material. Am. Rev. Resp. Dis. 107:784, 1973.

Knight, G., and K. Lichti. Comparison of cyclone and horizontal elutriator size selectors. Am. Ind. Hyg. Assoc. J. 31:437-441, 1970.

Kott, A. T., J. W. Gardner, R. S. Schecter, and W. DeGroot. The elasticity of pulmonary lung surfactants. J. Colloid Interface Sci. 47:265, 1974.

Krahl, V. Microstructure of the lung. Arch. Environ. Health 6:37, 1963.

LaBelle, C. W., and H. Brieger. Patterns and mechanisms in the elimination of dust from the lung. In: Inhaled Particles and Vapours. C. N. Davies, ed., Pergamon Press, Oxford, England, 1961. p. 356.

LaBelle, C. W., M. A. Bevilacqua, and H. Brieger. The influence of cigarette smoke on lung clearance. Arch. Environ. Health 12:588, 1966.

Landahl, H. D. On the removal of airborne droplets by the human respiratory tract. I. The lung. Bull. Math. Biophys. 12:43, 1950.

Landahl, H. D. Particle removel by the respiratory system. Bull. Math. Biophys. 25:29, 1963.

Landahl, H., and S. Black. Penetration of air-borne particulates through the human nose. J. Ind. Hyg. Toxicol. 29:269, 1947.

Landahl, H., and R. Herrmann. On the retention of air-borne particulates in the human lung. J. Ind. Hyg. Toxicol. 30:181, 1948.

Landahl, H. D., T. N. Tracewell, and W. H. Lassen. On the retention of airborne particulates in the human lung II. AMA Arch Ind. Health Occ. Med. 3:359, 1951.

Landahl, H. D., T. N. Tracewell, and W. H. Lassen. Retention of airborne particulates in the human lung III. AMA Arch. Ind. Hyg. Occup. Med. 6:508-511, 1952.

Lapp, N. L., J. L. Hankinson, H. Amandus and E. D. Palmes. Variability in the size of airspaces in normal human beings as estimated by aerosols. Thorax 30:293-299, 1975.

Larson, T. V., D. S. Covert, R. Frank, and R. J. Charlson. Ammonia in the human airways, neutralization of inspired acid sulfate aerosol. Science 197:161-163, 1977.

Lauweryns, J. M., and J. H. Baert. Alveolar clearance and the role of the pulmonary lymphatics. Am. Rev. Resp. Dis. 115:625, 1977.

Leach, L. J., E. A. Maynard, H. C. Hodge, J. K. Scott, C. L. Yuile, G. E. Sylvester, and H. G. Wilson. A five-year inhalation study with natural uranium-dioxide (UO_2) dust - I. Retention and biological effect in the monkey dog and rat. Health Phys. 18:599, 1970.

Leach, L. J., C. L. Yuile, H. C. Hodge, G. E. Sylvester, and H. B. Wilson. A five-year inhalation study with natural uranium dioxide (UO_2) dust - II. Postexposure retention and biological effects in the monkey, dog, and rat. Health Phys. 25:239, 1973.

Leeds, S. E., S. Reich, H. N. Uhley, J. J. Sampson, and M. Friedman. The pulmonary lymph flow after irradiation of the lungs of dogs. Chest 59:203, 1971.

Lever, J., cited in C. N. Davies. Deposition of aerosol in the human lung. In: Aerosole in Physik Medizin und Technik, Bad-Soden, W. Germany, Gesellschaft für Aerosolforschung, 1974, p. 90-99.

Lippmann, M. Deposition and clearance of inhaled particles in the human nose. Ann. Otol. Rhinol. Laryngol. 79:519-528, 1970.

Lippmann, M. Regional deposition of particles in the human respiratory tract. In: Handbook of Physiology, Section 9: Reactions to Environmental Agents. D. H. K. Lee, H. L. Falk, and S. D. Murphy, eds., The American Physiological Society, Bethesda, MD, 1977. pp. 213-232.

Lippmann, M. "Respirable" dust sampling. In: Air Sampling Instruments for Evaluation of Atmospheric Contaminants, 5th ed. American Conference of Governmental Industrial Hygienists, Cincinnati, OH, 1978. pp. G-1, G-23.

Lippmann, M., and R. Albert. The effect of particle size on the regional deposition of inhaled aerosols in the human respiratory tract. Am. Ind. Hyg. Assoc. J. 30:257, 1969.

Lippmann, M., R. E. Albert, and H. T. Peterson, Jr. The regional deposition of inhaled aerosols in man. In: Inhaled Particles III. W. H. Walton, ed., Unwin Brothers Limited, Surrey, England, 1971. p. 105.

Lippmann, M., M. S. Mok, and K. Wasserman. Anaesthetic management for children with alveolar proteinosis using extracorporeal circulation: a report of two cases. Br. J. Anaesth. 49:173-177, 1977.

Longley, M. Y. Pulmonary deposition of dust as affected by electric charges on the body. Am. Ind. Hyg. Assoc. J. 21:187, 1960.

Longley, M. Y., and C. M. Berry. Pulmonary deposition of aerosols: effect of electrostratic charging of the animal body and the aerosol. Arch. Environ. Health 2:533, 1961.

Lourenco, R. V., R. Loddenkemper, and R. W. Cargon. Patterns of distribution and clearance of aerosols in patients with bronchiectasis. Am. Rev. Resp. Dis. 106:857-866, 1972.

Luchsinger, P. G., B. LaGarde, and J. E. Kilfeather. Particle clearance from the human tracheobronchial tree. Am. Rev. Resp. Dis. 97:1046, 1968.

Luft, U. C. Spirometric methods. In: Aviation Medicine - Selected Reviews. C. S. White, W. R. Lovelace, F. G. Hirsch, eds., Pergamon Press, New York, NY, 1958. p. 168.

Lynch, J. R. Evaluation of size-selective presamplers: I. Theoretical cyclone and elutriator relationships. Am. Ind. Hyg. Assoc. J. 31:548-551, 1970.

Machlin, C. C. The alveoli of the mammalian lung: an anatomical study with clinical correlations. Proc. Inst. Med. 18:78, 1950.

Maguire, B. A., and D. Barker. A gravimetric dust sampling instrument (SIMPEDS): preliminary underground trials. Ann. Occup. Hyg. 12:197-201, 1969.

Marshall, R., and W. Holden. Changes in calibre of the smaller airways in man. Thorax 18:54, 1963.

Märtens, A., and W. Jacobi. Die In-Vivo Bestimmung der Aerosolteilchen-deposition im Atemtrakt bei Mund-Bzw. Nasenatmung. [In vivo determination of aerosol particle deposition in the total respiratory tract.] In: Aerosole in Physik, Medizin und Technik, Gesellschaft fur Aerosolforschung, Bad Soden, W. Germany, 1973. pp. 117-121.

Martin, D., and W. Jacobi. Diffusion deposition of small-sized particles in the bronchial tree. Health Phys. 23i:23-29, 1972.

McEuen, D. D., and J. L. Abraham. Particulate concentrations in pulmonary alveolar proteinosis. Environ. Res. 17:334-339, 1978.

Melandri, C., V. Prodi, G. Tarroni, M. Formignani, T. DeZaiacomo, G. R. Bompane, G. Maestri, and G. G. Giacomelli-Maltoni. On the deposition of unipolarly charged particles in the human respiratory tract. In: Inhaled Particles IV. W. H. Walton, ed., Pergamon Press, New York, NY, 1977. p. 193.

Melville, G. N. Changes in specific airway conductance in healthy volunteers following nasal and oral inhalation of SO_2. W. I. Med. J. 19:231-235, 1970.

Mercer, T. T. On the role of particle size in the dissolution of lung burdens. Health Phys. 13:1211, 1967.

Mercer, T. T. Aerosol Technology in Hazard Evaluation. Academic Press, New York, NY, 1973. pp. 66-280.

Michel, F. B., J. P. Marty, L. Quet and P. Cour. Penetration of inhaled pollen into the respiratory tract. Am. Rev. Resp. Dis. 115:609-616, 1977.

Miller, F. J., D. E. Gardner, J. A. Graham, R. E. Lee, Jr., W. E. Wilson, and J. D. Bachmann. Size considerations for establishing a standard for inhalable particles. J. Air Pollut. Control Assoc. 29:610-615, 1979.

Morgan, A., J. C. Evans, and A. Holmes. Deposition and clearance of inhaled fibrous minerals in the rat: studies using radioactive tracer techniques. In: Inhaled Particles IV. W. H. Walton, ed., Pergamon Press, Oxford, England, 1977. pp. 259-274.

Morgan, W. K. C. Industrial bronchitis. Br. J. of Ind. Med. 35:285-291, 1978.

Morgan, W. K. C., and A. Seaton. Occupational lung diseases. W. B. Saunders, Philadelphia, PA, 1975.

Morrow, P. E. Experimental studies of inhaled materials. Arch. Intern. Med. 126:466, 1970a.

Morrow, P. E. Models for the study of particle retention and elimination in the lung. In: Inhalation Carcinogenesis. M. G. Hanna, P. Nettesheim, J. R. Gilbert, eds., U.S. Atomic Energy Commission, Oak Ridge, TN, 1970b. p. 103.

Morrow, P. E. Lymphatic drainage of the lung in dust clearance. Ann. N.Y. Acad. Sci. 22:64-65, 1972.

Morrow, P. E. Alveolar clearance of aerosols. Arch. Intern. Med. 131:101, 1973.

Morrow, P. E., E. Mehrhof, L. Casarett, and D. Morken. An experimental study of aerosol deposition in human subjects. AMA Arch. Ind. Health 18:292, 1958.

Morrow, P. E., D. V. Pates, B. R. Fish, T. F. Hatch, and T. T. Mercer. International commission on radiological protection task group on lung dynamics, deposition and retention models for internal dosimetry of the human respiratory tract. Health Phys. 12:173-207, 1966.

Morrow, P. E., F. R. Gibb, and K. M. Gazioglu. A study of particulate clearance from the human lungs. Am. Rev. Resp. Dis. 96:1209, 1967a.

Morrow, P. E., F. R. Gibb, and K. M. Gazioglu. The clearance of dust from the lower respiratory tract of man: an experimental study. In: Inhaled Particles and Vapours. S. C. N. Davies, ed., Pergamon Press, Oxford, England, 1967b. p. 351.

Moss, O. R., and H. J. Ettinger. Respirable dust characteristics of polydisperse aerosols. Am. Ind. Hyg. Assoc. J. 31:546-547, 1970.

Muir, D. C., and C. N. Davies. The deposition of 0.5 m diameter aerosols in the lungs of man. Ann. Occup. Hyg. 10:161, 1967.

Nair, P. V. N., and V. G. Vohra. Growth of aqueous sulphuric acid droplets as a function of relative humidity. J. Aerosol Sci. 6:265, 1975.

National Academy of Sciences. Measurement and Control of Respirable Dust in Mines. National Academy of Sciences, Washington, DC, 1980. pp. 348-405.

Natusch, D. F. S. and J. R. Wallace. Urban aerosol toxicity: the influence of particle size. Science 186:695-699, 1979.

Natusch, D. F. S., J. R. Wallace, and C. A. Evans, Jr. Toxic trace elements: preferential concentration in respirable particles. Science 183:202-204, 1974.

Newhouse, M. T., M. Dolovich, G. Obminski, and R. K. Wolff. Effect of TLV levels of SO_2 and H_2SO_4 on bronchial clearance in exercising man. Arch. Environ. Health 33:24-32, 1978.

Niinimaa, V., P. Cole, S. Mintz and R. J. Shephard. The switching point from nasal to oronasal breathing. Resp. Physiol. 42:61-71, 1980.

Niinimaa, V., P. Cole, S. Mintz and R. J. Shephard. Oronasal distribution of respiratory airflow. Resp. Physiol. 43:69-75, 1981.

Oberdörster, G., F. R. Gibb, H. Beiter, S. T. Lu, P. E. Morrow. Studies of the lymphatic drainage of dog lungs. J. Toxicol. Environ. Health 4:571-586, 1978.

Olson, D. E., M. F. Sudlow, K. Horsfield, and G. F. Filley. Convective patterns of flow during inspiration. Arch. Intern. Med. 131:51-57, 1973.

Orenstein, A. J., ed. Proceedings of the Pneumonoconiosis Conference 1959, J & A Churchill, Ltd., London, England, 1960.

Owen, P. R. Turbulent flow and particle deposition in the trachea. In: Circulatory and Respiratory Mass Transport. G. E. W. Wolstenholme and J. Knight, eds., A CIBA Foundation Symposium. Little, Brown and Co., Boston, MA, 1969. pp. 236-252.

Pack, A., M. B. Hooper, W. Nixon, and J. C. Taylor. A computational model of pulmonary gas transport incorporating effective diffusion. Resp. Physiol. 29:101-124, 1977.

Paiva, M. Gas transport in the human lung. J. Appl. Physiol. 35:401, 1973.

Palmes, E. D., and C. S. Wang. An aerosol inhalation apparatus for human single breath deposition studies. Am. Ind. Hyg. Assoc. J. 32:43, 1971.

Palmes, E. D. and M. Lippmann. Influence of respiratory airspaces dimensions on aerosol deposition. Inhaled Particles IV. W. J. Walton, ed., Permagon Press, Oxford, England, 1977, pp. 127-135.

Pattle, R. E. The retention of gases and particles in the human nose. In: Inhaled Particles and Vapours. C. N. Davies, ed., Pergamon Press, Oxford, England, 1961a. p. 302.

Pattle, R. E. The lining complex of the lung alveoli. In: Inhaled Particles and Vapours. C. N. Davies, ed., Pergamon Press, Oxford, England, 1961b. p. 70.

Pavia, D., M. Thomson, and H. S. Shannon. Aerosol inhalation and depth of deposition in the human lung. Arch. Environ. Health 32:131, 1977.

Pavlik, I. The fate of light air ions in the respiratory pathways. Int. J. Biometeor. 11:175, 1967.

Pawley, J. B., and G. L. Fisher. Using simultaneous three colour X-ray mapping and digital-scan-stop for rapid elemental characterization of coal combustion by-products. J. Microsc. 110:87, 1977.

Pedley, T. J. A theory for gas mixing in a simple model of the lung. In: Fluid Dynamics of Blood Circulation and Respiratory Flow, Proceedings of Meeting, Naples, Italy, May 4-6, 1970 AGARD Conference Proceedings No. 65. Advisory Group for Aerospace Research and Development, Paris, France, 1970.

Pedley, T. J., R. C. Schroter, and M. F. Sudlow. Flow and pressure drop in systems of repeatedly branching tubes. J. Fluid Mech. 46:365-383, 1971.

Pfleger, R. C., and H. G. Thomas. Beagle dog pulmonary surfactant lipids. Arch. Intern. Med. 9:70, 1971.

Phalen, R. F., and P. E. Morrow. Experimental inhalation of metallic silver. Health Phys. 24:509-518, 1973.

Phalen, R. F., H. C. Yeh, G. M. Schum, and O. G. Raabe. Application of an idealized model to morphometry of the mammalian tracheobronchial tree. Anat. Rec. 190:167-176, 1978.

Polgar, G., and T. R. Weng. The functional development of the respiratory system from the period of gestation to adulthood. Am. Rev. Resp. Dis. 120:625-695, 1979.

Pratt, P. C. and K. H. Kilburn. Extent of pulmonary pigmentation as an indicator of particulate environmental air pollution. In: Inhaled Particles III. II. W. H. Walton, ed., Unwin Brothers Limited, The Gresham Press, Surrey, England, 1971, pp. 661-670.

Proctor, D. F., and D. L. Swift. The nose - a defense against the atmospheric environment. In: Inhaled Particles III. V. W. H. Walton, ed., Unwin Brothers, Limited, Surrey, England, 1971. p. 59.

Proctor, D. F., and H. N. Wagner. Clearance of particles from the human nose. Arch. Environ. Health 11:366, 1965.

Proctor, D. F., and H. N. Wagner. Mucociliary clearance in the human nose. In: Inhaled Particles and Vapours II. C. N. Davies, ed., Pergamon Press, Oxford, England, 1967. p. 25.

Proctor, D. F., D. L. Swift, M. Quinlan, S. Salman, Y. Takagi, and S. Evering. The nose and man's atmospheric environment. Arch. Environ. Health 18:671, 1969.

Proctor, D. F., I. Andersen, and G. Lundquist. Clearance of inhaled particles from the human nose. Arch. Intern. Med. 131:132, 1973.

Pruitt, K. M., M. J. Cherny, and H. L. Spitzer. Physical and chemical characterization of pig lung surfactant lippoprotein. Arch. Intern. Med. 9:6, 1971.

Pump, K. K. The morphology of the finer branches of the bronchial tree of the human lung. Dis. Chest 46:379, 1964.

Raabe, O. G. Some important consideration in use of power function to describe clearance data. Health Phys. 13:293, 1967.

Raabe, O. G. Particle size analysis utilizing grouped data and the log-normal distribution. Aerosol Sci. 2:289, 1971.

Raabe, O. G. Aerosol aerodynamic size conventions for inertial sampler calibration. J. Air Pollut. Control Assoc. 26:856, 1976.

Raabe, O. G. Deposition and Clearance of Inhaled Aerosols. U.S. Department of Energy. National Technical Information Service, UCD-472-503, Springfield, VA, 1979.

Raabe, O. G., and M. Goldman. A predictive model of early mortality following acute inhalation of PuO_2 aerosols. Radiat. Res. 78:264-277, 1979.

Raabe, O. G., H. C. Yeh, G. M. Schum, and R. F. Phalen. Tracheobronchial Geometry: Human, Dog, Rat, Hamster, LF-53. Lovelace Foundation, Albuquerque, NM, 1976.

Raabe, O. G., H. C. Yeh, G. J. Newton, R. F. Phalen, and D. J. Velasquez. Deposition of inhaled monodisperse aerosols in small rodents. In: Inhaled Particles IV. W. H. Walton, ed., Pergamon Press, New York, NY, 1977. p. 3-22.

Raabe, O. G., S. V. Teague, N. L. Richardson, and L. S. Nelson. Aerodynamic and dissolution behavior of fume aerosols produced during the combustion of laser-ignited plutonium droplets in air. Health Phys. 35:663, 1978.

Ramsden, D., M. E. D. Bains, and D. C. Fraser. In vivo and bioassay results from two contrasting cases of plutonium-239 inhalation. Health Phys. 19:9, 1970.

Reifenath, R. Chemical analysis of the lung alveolar surfactant obtained by alveolar micropuncture. Resp. Physiol. 19:35, 1973.

Rudolf, G., and J. Heyder. Deposition of aerosol particles in the human nose. In: Aerosole in Naturwissenschaft, Medizin und Technik. V. Böhlan, ed. Proceedings of a Conference held in Bad Soden, October 16-19, 1974. Gesellschaft für Aerosolforschung, Bad Soden, West Germany, 1974.

Saibene, F., P. Mognoni, C. L. LaFortuna, and R. Mostardi. Oronasal breathing during exercise. Pflugers Arch. J. d. Physiol. 378:65-69, 1978.

Sanchis, J., M. Dolovich, R. Chalmers, and M. Newhouse. Quantitation of regional aerosol clearance in the normal human lung. J. Appl. Physiol. 33:757, 1972.

Sanders, C. L., and R. R. Adee. Phagocytosis of inhaled plutonium oxide - ^{239}Pu particles by pulmonary macrophages. Science 162:918-920, 1968.

Scarpelli, E. M. The Surfactant System of the Lung. Lea and Febiger, Philadelphia, PA, 1968.

Scherer, P. W., L. H. Shendalman, and N. M. Greene. Simultaneous diffusion and convection in a single breath lung washout. Bull. Math. Biophys. 34:393-412, 1972.

Scherer, P. W., L. H. Shendalman, N. M. Greene, and A. Bouhuys. Measurement of axial diffusivities in a model of the bronchial airways. J. Appl. Physiol. 38:719-723, 1975.

Scherer, P. W., F. R. Haselton, L. M. Hanna, and D. R. Stone. Growth of hygroscopic aerosols in a model of bronchial airways. J. Appl. Physiol.: Resp. Environ. Exercise Physiol. 47:544-550, 1979.

Schlesinger, R. B. Mucociliary interaction in the tracheobronchial tree and environmental pollution. Biol. Sci. 23:567, 1973.

Schlesinger, R. B., and M. Lippmann. Particle deposition in the trachea: in vivo and in hollow casts. Thorax 31:678, 1976.

Schlesinger, R. B., and M. Lippmann. Selective particle deposition and bronchogenic carcinoma. Environ. Res. 15:424, 1978.

Schroter, R. C., and M. F. Sudlow. Flow patterns in models of the human bronchial airways. Resp. Physiol. 7:341, 1969.

Shanty, F. Deposition of Ultrafine Aerosols in the Respiratory Tract of Human Volunteers. Ph.D. thesis, Johns Hopkins University, Baltimore, MD, 1974.

Sherwin, R. P., M. L. Barman, and J. L. Abraham. Silicate pneumoconiosis of farm workers. Lab. Invest. 40:576-582, 1979.

Sherwood, T. K., R. L. Pigford, and C. R. Wilke. Mass Transfer. McGraw-Hill, New York, NY, 1975. pp. 13, 677.

Silverman, L., and C. E. Billings. Pattern of airflow in the respiratory tract. In: Inhaled Particles and Vapours. C. N. Davies, ed., Pergamon Press, Oxford, England, 1961. p. 9.

Smith, F. A., and E. A. Boyden. An analysis of the segmental bronchi of the right lower lobe of fifty injected lungs. J. Thoroc. Surg. 18:195, 1949.

Snipes, M. B., and M. F. Clem. Retention of microspheres in the rat lung after intratracheal instillation. Environ. Res. 24:33-41, 1981.

Speizer, F. E., and N. R. Frank. The uptake and release of SO_2 by the human nose. Arch. Environ. Health 12:725-728, 1966.

Spiegelman, J. R., G. D. Hanson, A. Lazarus, R. J. Bennett, M. Lippmann, and R. E. Albert. Effect of acute SO_2 exposure on bronchial clearance in the donkey. Arch. Environ. Health 17:321-326, 1968.

Stahl, W. R. Scaling of respiratory variables in mammals. J. Appl. Physiol. 22:453-460, 1967.

Stahlhofen W., J. Gebhart, and J. Heyder. Experimental determination of the regional deposition of aerosol particles in the human respiratory tract. Am. Ind. Hyg. Assoc. J. 41:385-398a, 1980.

Stahlhofen, W., J. Gebhart and J. Heyder. Biological variability of regional deposition of aerosol particles in the human respiratory tract. Am. Ind. Hyg. Assoc. J. 42:348-352, 1981.

Stockham, J. D., and E. G. Fochtman, eds. Particle Size Analysis. Ann Arbor Science, Ann Arbor, MI, 1979. pp. 140.

Strandberg, L. G. SO_2 absorption in the respiratory tract. Arch. Environ. Health 9:160-166, 1964.

Strecker, F. J. Tissue reactions in rat lungs after dust inhalation with special regard to bronchial dust elimination and to the penetration of dust into the lung interstices and lymphatic nodes. In: Inhaled Particles and Vapours II. C. N. Davies, ed., Pergamon Press, Oxford, England. 1967. p. 141.

Sweet, D. V., W. E. Crouse, and J. V. Crable. Chemical and statistical studies of contaminants in urban lungs. Am. Ind. Hyg. Assoc. J. 39: 515-526, 1978.

Swift, D. L., F. Slanty, and J. T. O'Neil. Human respiratory deposition patterns of fume-like particles. Presented in part at the 1977 American Industrial Hygiene Conference, New Orleans, LA, May 26, 1977.

Taplin, G. V., N. D. Poe, E. K. Dore, A. Greenberg, and T. Isawa. Radioaerosol inhalation scanning. In: Pulmonary Investigation with Radionuclides. W. M. Smoals, ed., Charles C. Thomas, Springfield, IL, 1970. pp. 296-317.

Tarroni, G., C. Melandri, V. Prodi, T. deZaiacomo, M. Formignani, and P. Bassi. An indication on the biological variability of aerosol total deposition in humans. Am. Ind. Hyg. Assoc. J. 41:826-831, 1980.

Taulbee, D. B., and C. P. Yu. A theory of aerosol deposition in the human respiratory tract. J. Appl. Physiol. 38:77, 1975.

Taulbee, D., C. Yu, and J. Heyder. Aerosol transport in the human lung from analysis of single breaths. J. Appl. Physiol. 44:803, 1978.

Tenney, S. M., and J. E. Remmers. Comparative quantitative morphology of the mammalian lung: diffusing area. Nature 197:54, 1963.

Tenney, S. M., and D. Bartlett. Comparative quantitative morphology of the mammalian lung: trachea. Resp. Physiol. 3:130, 1967.

Thomas, R. G. Transport of relatively insoluble materials from lung to lymph nodes. Health Phys. 14:111, 1968.

Threshold Limits Committee. Threshold Limit Values of Air Borne Contaminants for 1968. American Conference of Governmental Industrial Hygienists, Cincinnati, OH, 1968.

Thurlbeck, W. M., and J. B. Haines. Bronchial dimensions and stature. Am. Rev. Resp. Dis. 112:142, 1975.

Tuttle, W. C., and S. C. Westerberg. Alpha-1-globulin trypsin inhibitor in canine surfactant protein. Proc. Soc. Exp. Biol. Med. 146:232, 1974.

Uddströmer, M. Nasal respiration. Acta Otolaryngol. Suppl. 42:3-146, 1940.

Van As, A., and I. Webster. The organization of ciliary activity and mucus transport in pulmonary airways. S. A. Med. J. 46:347, 1972.

Van Ree, J. H. L., and H. A. E. van Dishoeck. Some investigations on nasal ciliary activity. Pract. Otorhinolaryng (Basel) 24:383, 1962.

Van Wijk, A. M., and H. S. Patterson. The percentage of particles of different sizes removed from dust-laden air by breathing. J. Ind. Hyg. Toxicol. 22:31, 1940.

Verzar, F., J. Keith, and V. Parchet. Temperatur und Feuchtigkeit der Luft in den Atemwegen. [Temperature and moisture content of the air in the respiratory tract.] Pfluegers Arch. J. d. Physiol. 257:400, 1953.

Waligora, S. J., Jr. Pulmonary retention of zirconium oxide (^{65}Nb) in man and beagle dogs. Health Phys. 20:89, 1971.

Walkenhorst, W. Untersuchungen an einem nach teilchengrossen geordneten Mischstaub im atembaren korngrossenbereich. In: Inhaled Particles and Vapours II. C. N. Davies, ed., Pergamon Press, Oxford, England, 1967. p. 563.

Wang, C. S. Gravitational deposition from laminar flows in inclined channels. J. Aerosol Sci. 6:19, 1975.

Washburn, E. W., ed. National Research Council of the U.S.A. International Critical Tables of Numerical Data, Physics, Chemistry and Technology, vol. 3. McGraw-Hill, New York, NY, 1928.

Weibel, E. R. Morphometry of the Human Lung. Academic Press, New York, NY, 1963.

West, J. B. Observations on gas flow in the human bronchial tree. In: Inhaled Particles and Vapours. C. N. Davies, ed., Proceedings of an International Symposium organized by the British Occupational Hygiene Society, 1960. Pergamon Press, New York, NY, 1961. pp. 3-7.

West, J. B. Respiratory Physiology: the Essentials. Williams and Wilkins, Baltimore, MD, 1974.

West, J. B. Respiratory Physiology: the Essentials. Williams and Wilkins, Baltimore, MD, 1977.

Whimster, W. F. The microanatomy of the alveolar duct system. Thorax 25:141, 1970.

Whitby, K. T. The physical characteristics of sulfur aerosols. Atmos. Environ. 12:135, 1978.

Wilson, T. A., and K. Lin. Convection and diffusion in the airways and the design of the bronchial tree. In: Airway Dynamics; Physiology and Pharmacology. A. Bouhuys, ed., Charles C. Thomas, Springfield, IL, 1970. pp. 5-19.

Wolff, R. K., M. Dolovich, C. M. Rossman, and M. T. Newhouse. Sulfur dioxide and tracheobronchial clearance in man. Arch. Environ. Health 30:521-527, 1975.

Yeates, D. B., N. Aspin, H. Levinson, M. T. Jones, and A. C. Bryan. Mucociliary trachael transport rates in mon. J. of Appl. Physiol. 39:487-495, 1975.

Yeh, H. C. Use of a heat transfer analogy for a mathematical model of respiratory tract deposition. Bull. Math. Biol. 36:105, 1974.

Yeh, H. C., and G. M. Schum. Models of human lung airways and their application to inhaled particle deposition. Bull. Math. Biol. 42:461-480, 1980.

Yeh, H. C., R. F. Phalen, and O. G. Raabe. Factors influencing the deposition of inhaled particles. Environ. Health Perspect. 15:147, 1976.

Yu, C. P. An equation of gas transport in the lung. Resp. Physiol. 23:257-266, 1975.

Yu, C. P. Precipitation of unipolarly charged particles in cylindrical and spherical vessels. J. Aerosol Sci. 8:237, 1977.

Yu, C. P. A two component theory of aerosol deposition in human lung airways. Bull. Math. Biol. 40:693-706, 1978.

Yu, C. P., P. Nicolaides, and T. T. Soong. Effect of random airway sizes on aerosol deposition. Am. Ind. Hyg. Assoc. J. 40:999-1005, 1979.

Zenz, C., ed. Occupational Medicine: Principles and Practical Applications. Year Book Medical Publishing, Inc., Chicago, IL, 1975. pp. 113-115.

12. TOXICOLOGICAL STUDIES

12.1 INTRODUCTION

This chapter describes the toxicity of sulfur oxides (SO_x) and particulate matter (PM) in animals. The health effects of SO_x and PM have also been reviewed by the National Academy of Sciences National Research Council (1978, 1979). The toxic effects of SO_x and of atmospheric aerosols overlap because the salts of sulfuric acid (ammonium sulfate, sodium sulfate, and related compounds) are significant components of atmospheric aerosols (see Chapter 2). The toxicology of all forms of SO_x must be considered as a whole. For example, in the ambient air, sulfur dioxide (SO_2) may interact with aerosols, be absorbed on particles, or be dissolved in liquid aerosols. To a lesser degree, similar interactions may occur in the air within the respiratory tract. Sulfuric acid (H_2SO_4) aerosols may react with ammonia, forming ammonium sulfate [$(NH_4)_2SO_4$] and ammonium bisulfate (NH_4HSO_4) in the ambient air, in the animal exposure chamber atmosphere before inhalation, or, to a lesser degree, in the respiratory tract simultaneously upon inhalation (see Section 12.3). Biological interactions can also occur, resulting in a mixture of pollutants that has additive, synergistic, or antagonistic health effects compared to the effects of the single pollutants.

Discussions of the deposition and clearance of SO_x are limited here (see Chapter 11 for more details). The major toxic effects of sulfur compounds, whether caused by SO_2, H_2SO_4, or some sulfate salts, include immediate irritation of the respiratory tract. Most measurements of this irritation have been made through studies of the respiratory mechanics of the experimental animal. Similar studies of respiratory mechanics have been done with human subjects either experimentally or environmentally exposed. The general effects of SO_2 on the respiratory mechanics of animals and man are the same. The animal studies reviewed here present some details of the metabolism of SO_2 and bisulfite, the effects of SO_2 on the biochemistry, physiology, and morphology of the respiratory tract, and the potential effects on organs other than the lung.

The physicochemical diversity of PM in the atmosphere is a major problem. When the local concentration of any organic compound exceeds its vapor pressure, it will condense and be found in the particulate fraction when sampled. Some may be sorbed on the surfaces of inorganic particles, which can have a wide chemical variety. Progress is being made toward a better understanding of the toxicity of these materials associated with particles, but at present inadequate data are available. A more complete treatment of the health effects of polycyclic organic matter is found in a review by the Environmental Criteria and Assessment Office of EPA (1978), which gives an overview also for some of the heavy metals present in polluted air. More detail is found in documents dealing specifically with each heavy metal (Environmental Criteria and Assessment Office, 1979; Office of Research and Development, 1977; Committee on Biologic Effects of Atmospheric Pollutants, Vanadium, 1974; Committee on Medical and Biologic Effects of Environmental Pollutants, Nickel, 1975; Committee on Biologic

Effects of Atmospheric Pollutants, Lead, 1972; Committee on Biologic Effects of Atmospheric Pollutants, Chromium, 1974; Committee on Medical and Biologic Effects of Environmental Pollutants, Arsenic, 1977; National Academy of Sciences, Iron, 1979; National Academy of Sciences, Zinc, 1979).

Interactions between SO_x and other pollutants are reviewed briefly because of the sparsity of the data available. Some of these studies are controversial and have not been duplicated, especially those dealing with the potential mutagenic effects of SO_2 and the interaction of SO_2 with known carcinogens.

Another difficulty with a review of the toxicology of SO_x is the sparsity of recent studies. Since the early 1970s few studies have appeared and work in progress is not included here. As a result, a certain degree of sophistication is lacking in some of the intrepretations, not through a lack of appreciation of the problem, but simply because insufficient information is available.

12.2 EFFECTS OF SULFUR DIOXIDE

12.2.1 Biochemistry of Sulfur Dioxide

This section largely covers *in vitro* experiments, in which the potential target (i.e., cells, enzymes, other molecules, etc.) is exposed to the toxicant outside the body. The potential problems of such a system include the fact that some homeostatic or repair mechanisms are absent or that pollutants sometimes act by indirect mechanisms. For example, the pollutant affects target A that in turn alters target B. Thus, if only target B were present, the effect would not be observed. In addition, the dosimetric relationships of *in vitro* studies to *in vivo* studies have not been defined. Therefore, effective concentrations cannot be extrapolated directly from *in vitro* to *in vivo* studies. For the above reasons, there is some controversy as to whether observed *in vitro* reactions can be extrapolated to *in vivo* mechanisms of toxicity. Nonetheless, sound *in vitro* investigations can show whether a given pollutant has the potential of affecting a given target. *In vitro* studies are best used to provide guidance for *in vivo* investigations or when *in vivo* results have been observed. In the latter case, the relatively simplified *in vitro* systems can sometimes elucidate the potential mechanisms of toxicity. To these ends, they can be useful.

Knowledge of the chemistry of sulfurous acid and SO_2 is necessary to understand the physiological and toxicological properties of SO_2. Sulfur dioxide is the gaseous anhydride of sulfurous acid. It dissolves readily in water, and at physiological pH near neutrality, hydrated SO_2 readily dissociates to form bisulfite and sulfite ions as illustrated by Equations 12-1 and 12-2. The rate of hydration of SO_2 is very rapid, with a rate constant (k_1) of $3.4 \times 10^6\ M^{-1}\ sec^{-1}$; the rate constant of the reverse reaction is $2 \times 10^8\ M^{-1}\ sec^{-1}$ at 20°C (Equation 12-1) (Tartar and Garetson, 1941). The logarithms of the inverse of the dissociation constants (pK_a) of sulfurous acid are 1.37 and 6.25 (in dilute salt solutions) (Tartar and Garetson, 1941); consequently, at pH 7.4, the concentration of sulfite ions is about 14 times that of bisulfite, but in rapid equilibrium. Hence, SO_2 can be treated as bisulfite/sulfite and conversely.

$$SO_2 + H_2O \underset{k_2}{\overset{k_1}{\rightleftharpoons}} H_2SO_3 \qquad 12\text{-}1$$

$$H_2SO_3 \overset{pK_a = 1.37}{\rightleftharpoons} H^+ + HSO_3^- \overset{pK_a = 6.25}{\rightleftharpoons} H^+ + SO_3^= \qquad 12\text{-}2$$

12.2.1.1 Chemical Reactions of Bisulfite with Biological Molecules--Sulfur dioxide reacts readily with all major classes of biomolecules. Reactions of SO_2 or bisulfite with nucleic acids, proteins, lipids, and other biological components have been repeatedly demonstrated *in vitro*. There are three important reactions of bisulfite with biological molecules: sulfonation, production of free radicals by autooxidation, and addition to cytosine.

Sulfonation, or sulfitolysis, (Gilbert, 1965) results from the nucleophilic attack of bisulfite on disulfides:

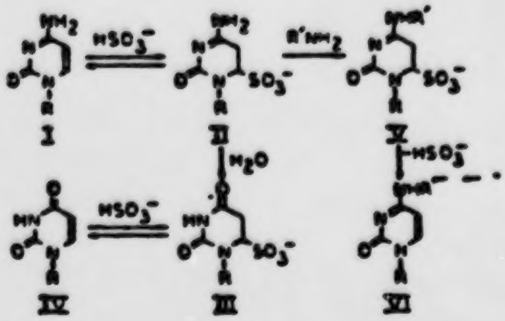
$$RSSR' + HSO_3^- \rightleftharpoons RSSO_3^- + R'SH \qquad 12\text{-}3$$

The reaction produces S-sulfonates ($RSSO_3^-$) and thiols (R'SH). Gunnison and Benton (1971) and Gunnison and Palmes (1973) provided direct evidence for the formation of plasma S-sulfonates *in vivo*. Any plasma protein containing a disulfide group could react to form an S-sulfonate. Small molecular weight disulfides, such as oxidized glutathione, can also be reactants. Generally, analyses of plasma S-sulfonates have been restricted to diffusable (dialyzable or small molecular weight compounds) and nondiffusable (nondialyzable or protein) S-sulfonates. The exact molecular species have not been determined, and the results of these analyses represent pools of the two groups of compounds. S-sulfonates can react with thiols, either reduced glutathione or protein thiol groups, to form sulfite and disulfide. Since this reverse reaction is facile, it is hypothesized that S-sulfonates are transportable forms of bisulfite within the body. Sulfitolysis therefore represents both a mechanism of toxicity and means of detoxification and redistribution of a reactive molecule, bisulfite.

Similar reversible nucleophilic addition of bisulfite to a variety of biologically important molecules has been reported, but the toxicological importance of these chemical species is uncertain. It is not likely, for example, that the reactions of bisulfite with pyrimidine nucleotides (NAD or NADP), reducing sugars, or thiamine are important to the toxicity of bisulfite or SO_2.

Autooxidation of bisulfite occurs through a multistep chain reaction (Hayon et al., 1972; Backstrom, 1927; Fridovich and Handler, 1958, 1960; Asada and Kiso, 1973; Peiser and Yang, 1977; Yip and Hadley, 1966; Rotilio et al., 1970; Nakamura, 1970; Klebanoff, 1961; Yang, 1967; McCord and Fridovich, 1969a,b). These reactions may be important because they produce hydroxyl (·OH) and superoxide ($\cdot O_2^-$) free radicals as well as singlet oxygen ($\cdot O_2$). These chemical species of oxygen, which are highly reactive and also produced by ionizing radiation, are theoretically responsible for the lethal effects of ionizing radiation. Autooxidation of bisulfite could lead to increased concentrations of these reactive chemical species within the cell and hypothetically could lead to similar adverse effects. The reactive forms of oxygen can also initiate peroxidation of the lipid bilayer of cells. Peroxidation of cellular lipids, especially plasma membrane lipids, is thought to be highly deleterious (Kaplan et al., 1975). No direct evidence has been presented, however, to support peroxidation of cellular lipids as a mechanism of toxicity of SO_2.

Bisulfite addition to cytosine results in the formation of uracil. The reaction of bisulfite with nucleic acids are as follows (Shapiro and Weisgras, 1970; Shapiro et al., 1970a,b; Hayatsu, 1976):



Changes introduced by bisulfite reaction at specific locations in the genome were shown to produce mutants in SV40 with the expected DNA sequence change after replication in vivo (Shortle and Nathans, 1978).

While the respiratory effects of SO_2 may be due to sulfonation (Alarie, et al, 1973d), the fact that all disulfides in the respiratory tract will likely undergo this reaction means that no single protein or small molecular weight compound can presently be identified as the target or receptor for SO_2 in this toxic lesion.

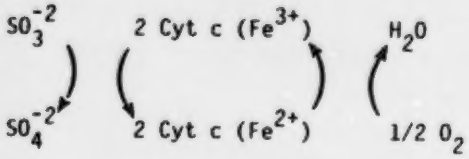
12.2.1.2 Metabolism of Sulfur Dioxide

12.2.1.2.1 Integrated metabolism. There are several studies of the metabolism of exogenously supplied SO_2, sulfite, or bisulfite. While quantitative differences exist between inhaled SO_2 and ingested bisulfite with regard to the rate of clearance of plasma S-sulfonates, one of the key intermediates in sulfite metabolism (Gunnison and Palmes, 1973), no qualitative differences exist in the metabolism of inhaled SO_2 and injected or ingested bisulfite or sulfite.

The importance of the appearance of plasma S-sulfonates ($RSSO_3^-$) lies in their potential ability to serve as a circulating pool of sulfite molecules (Gunnison and Palmes, 1973) as evidenced by the presence of ^{35}S from $^{35}SO_2$ in nonpulmonary tissues such as ovaries (Frank et al., 1967). Continuous inhalation of 26.2 mg/m^3 (10 ppm) SO_2 resulted in 38 ± 15 nmole of plasma S-sulfonates/ml in rabbits after about 4 days (Gunnison and Palmes, 1973). The clearance of plasma S-sulfonates generated by either inhalation of SO_2 or ingestion of sulfite in the drinking water was exponential, exhibiting only a single compartment in most rabbits. The half-life was 4.1 days for S-sulfonates generated by inhalation vs. 1.3 days for those generated by ingestion (Gunnison and Palmes, 1973). The mechanism for this quantitative difference in clearance rates has not yet been found.

Inhaled SO_2 quickly penetrates the nasal mucosa and airways as shown by the rapid appearance of ^{35}S in the venous blood of dogs inhaling $^{35}SO_2$ (Yokoyama et al., 1971). A significant fraction of the blood ^{35}S was probably in the form of plasma S-sulfonates. Most of the inhaled SO_2 is presumed to be detoxified by the sulfite oxidase pathway (predominantly in the liver but also in other organs), forming sulfate, which is excreted in the urine. The dominance of this reaction has been supported by studies of sulfite oxidase inhibition (Cohen, et al. 1972) that are discussed below, and by the appearance of about 85 percent of the inhaled $^{35}SO_2$ as urinary sulfate in dogs (Yokoyama et al., 1971). A small fraction (10 to 15 percent) of the urinary ^{35}S was in the form of H_2SO_4 esters. Sulfate arising from the oxidation of sulfite can enter the sulfate pool and be incorporated into sulfate macromolecules including glycosaminoglycans and glycoproteins. These macromolecules are actively synthesized by the respiratory mucosa and may account for the presence of radiolabeled sulfur in the respiratory tract following inhalation of $^{35}SO_2$ (Yokoyama et al., 1971). Most of the nondialyzable ^{35}S detected by Yokoyama et al. (1971) was bound to the α-globulin fraction of plasma. The chemical form of the ^{35}S was not determined. Yokoyama et al. (1971) speculated that the ^{35}S present in the α-globulin fraction was in the form of sulfonated carbohydrates. The problem needs further clarification. According to Gunnison and Palmes (1973), plasma S-sulfonated proteins may also have contained the ^{35}S. They have suggested that the slow clearance of plasma S-sulfonates is an important factor in determining toxicity, but have not reported intracellular levels of S-sulfonates or sulfite.

12.2.1.2.2 Sulfite oxidase. The biochemistry of sulfite oxidase is important as a mechanism of detoxification of sulfite. Sulfite oxidase is a metallo-hemo protein with molybdenum and protoheme as the prosthetic groups (Cohen et al., 1972). It exists in animals (Cohen et al., 1972,1974; Howell and Fridovich, 1968; Cohen and Fridovich, 1971a,b; Wattiaux-DeConinck and Wattiaux, 1974), bacteria (Lyric and Suzuki, 1970), and plants (Tager and Rautanen, 1955; Arrigoni, 1959; Fromageot et al., 1960). In both plants and animals, the enzyme is located in the mitochondria. Purified sulfite oxidase can utilize either cytochrome c or oxygen as the electron acceptor (Cohen and Fridovich, 1971a). When coupled with cytochrome c to the mitochondrial respiratory chain, sulfite oxidase reduces molecular oxygen to water (Equation 12-4), whereas during oxygen reduction, the product formed is hydrogen peroxide (Equation 12-5).



12-4

$$SO_3^{-2} + H_2O + O_2 \rightarrow SO_4^{-2} + H_2O_2 \qquad 12\text{-}5$$

Direct reduction of molecular oxygen by sulfite oxidase is prevented in the presence of ferric cytochrome c. In intact mitochondria, therefore, sulfite oxidation occurs through the interaction of sulfite oxidase with the respiratory chain of the mitochondria, producing 1 mole of ATP/mole of sulfite oxidized.

In three reported cases in humans, a rare genetic defect in sulfite oxidase resulted in severe neurological problems (Mudd et al., 1967; Irreverre et al., 1967; Shih et al., 1977; Duran et al., 1979). Dietary factors can, however, alter the enzymatic activity. Because sulfite oxidase requires molybdenum, Cohen et al. (1973) were able to deplete rats of sulfite oxidase by feeding them a low molybdenum diet and treating them with 100 ppm of sodium tungstate in drinking water. Tungsten competes with molybdenum and essentially abolishes the activity of sulfite oxidase and xanthine oxidase, the two major molybdo-proteins of rat liver. Similar decreases were observed in the lung and other organs. The LD_{50} for interperitoneally injected bisulfite was found to be 181 mg $NaHSO_3$/kg in the sulfite oxidase-deficient rats compared to 473 mg/kg in the nondeficient animals.

Attempts to induce higher levels of sulfite oxidase through pretreatment of the rats with SO_2/bisulfite or phenobarbital failed (Cohen et al., 1973). Since sulfite oxidase is a mitochondrial enzyme with a long half-life, it is not likely that phenobarbital or chronic exposure to SO_2 would result in adaptation through induction of higher levels of sulfite oxidase.

12.2.1.3 Activation and Inhibition of Enzymes by Bisulfite--Both inhibition and activation of specific enzymes have been reported. This may be due to formation of S-sulfonates, since disulfide bonds often stabilize the tertiary structure of proteins. Sulfite ions activated several phosphatases including ATP-ase (Marunouchi and Mori, 1967) and 2,3-diphosphoglyceric acid phosphatase (Harkness and Roth, 1969). The mechanism by which activation occurs is unknown. Inhibition of several enzymes has also been reported, including aryl sulfatase (Harkness and Roth, 1969), choline sulfatase (Takebe, 1961), rhodanase (Lyric and Suzuki, 1970), and hydroxyl amine reductase (Zucker and Nason, 1955). Malic dehydrogenase was inhibited by micromolar concentrations of bisulfite (Wilson, 1968; Ziegler, 1974). Other dehydrogenases (Oshino and Chance, 1975) and flavoprotein oxidases are inhibited by bisulfite.

Bisulfite effectively inhibits a number of other enzymes, including potato and rabbit muscle phosphorylase (Kamogawa and Fukui, 1973). Bisulfite inhibition was competitive with respect to glucose-1-phosphate and inorganic phosphate, suggesting that the bisulfite inhibition was caused by competition of bisulfite with the phosphate binding site of phosphorylase. Several important coenzymes (such as pyridoxylphosphate, NAD, NADP, FMN, FAD, and folic acid) may react with sulfite to form additional products as discussed above. As a result, these coenzymes could theoretically aid in inhibiting a wide variety of critical enzymic reactions. Pyridine coenzyme-bisulfite adduct (Tuazon and Johnson, 1977) and flavoenzyme-bisulfite adduct (Muller and Massey, 1969; Massey et al., 1969), which have been studied in detail, have been shown to be biologically inactive.

Despite all of the data obtained using _in vitro_ systems on the inhibition of enzymes by bisulfite/SO_2, no inhibition or activation has been determined _in vivo_ with SO_2 exposure. Such inhibition may occur, but there has been no concerted effort to search for inhibition of specific enzymes during SO_2 exposure.

12.2.2 Mortality

The acute lethal effects of SO_2 have been examined mostly in the older literature and have been reviewed in the previous Air Quality Criteria Document for Sulfur Oxides (National Air Pollution Control Administration, 1970). In early studies, several different animal species were examined for susceptibility to SO_2. These data show that neither rats nor mice died at exposures of 65.5 mg/m^3 (25 ppm) for up to 45 days, a conclusion confirmed by Laskin et al. (1970). Mortality could be associated with long-term exposure to SO_2 at 134 mg/m^3 (51 ppm) or higher. The clinical signs of SO_2 intoxication appear to vary with the dose rate (Cohen et al., 1973). At concentrations below approximately 1,310 mg/m^3 (500 ppm), mortality is associated with respiratory insufficiency; above this concentration, mortality is ascribed to central nervous disturbances producing seizures and paralysis of the extremities. These clinical signs depend upon the presence and activity of sulfite oxidase, as Cohen et al. (1973) observed shorter survival times and higher mortality rates in tungsten-treated animals. Injections of histamine or adrenalectomy can increase the lethality of SO_2 (Leong et al., 1961).

Matsumura (1970a,b) examined the effect of a 30-min exposure to several air pollutants on mortality consequent to the anaphylactic response of guinea pigs to protein antigens. Sensitization to the antigen administered by aerosol was augmented by pretreatment with 786 mg/m^3 (300 ppm) SO_2, but not with 472 mg/m^3 (180 ppm).

On the basis of mortality due to acute exposure, SO_2 is far less toxic than ozone and is similar in toxicity to nitrogen dioxide. Concentrations required to produce mortality from SO_2 are far in excess of those that occur in the atmosphere due to pollution (Table 12-1).

12.2.3 Morphological Alterations

Because of the high solubility of SO_2 in water and biological fluids, morphological and physiological effects occur mostly in the upper airways; however, changes have also been detected in the lower airways (Table 12-2). At the relatively high concentrations (>26.2 mg/m^3; 10 ppm) used in most studies designed to detect morphological alterations, most of the inhaled SO_2 is removed by the extrathoracic (nasopharyngeal) cavity. (See Chapter 11, Section 11.2.4, for an expanded discussion of SO_2 absorption.) In rabbits, the concentration of inspired SO_2 determines how much is removed in the ET cavity as opposed to the tracheobronchial (TB) and pulmonary (P) regions of the lung (Strandberg, 1964). At SO_2 concentrations greater than 26.2 mg/m^3 (10 ppm), 90 to 95 percent is removed in the ET cavity. A small part, 3 to 5 percent, is removed by the TB-P region. Thus, in this range, most of the dose is delivered to the nasal turbinates with only a small percentage going to the lung parenchyma. At lower concentrations of inspired SO_2, [such as 0.13 mg/m^3 (0.05 ppm)] which are closer to ambient levels, only 40 percent of the dose is absorbed by the ET cavity upon inspiration, while another 40 percent is removed by the respiratory tract upon expiration. Thus, at lower concentrations, the actual percentage of SO_2 removed in specific regions of the respiratory tract is not known precisely. In the dog, over 95 percent is removed by the upper airways and nose at concentrations between 2.62 and 131 mg/m^3 (1 and 50 ppm) SO_2 (Frank et al., 1967, 1969). A more detailed consideration of SO_2 extraction by airways is given in Section 12.2.4 below.

Giddens and Fairchild (1972) pointed out that these differences in removal of inspired SO_2 could explain the apparent anomaly of little damage to the lower respiratory tract at high SO_2 concentrations. They studied the effects of inhaled SO_2 on the nasal mucosa of mice. Two groups of mice were used; one group that was free of specific upper respiratory pathogens, and a conventional group that was presumed to be infected or to have a latent infection of upper respiratory pathogens. The mice were exposed continuously to 26.2 mg/m^3 (10 ppm) SO_2 for a maximum of 72 hours. Pathological changes in the nasal mucosa appeared after 24 h of exposure and increased in severity after 72 h of exposure. Mice free of upper respiratory pathogens had fewer pathological findings than the conventionally raised animals. Giddens and Fairchild (1972) concluded that resident or acquired pathogens exacerbated the morphological changes. These alterations were, however, qualitatively identical in both groups of animals. Cilia were lost from the nasal mucosa, vacuolization appeared, the mucosa decreased to about one half the normal thickness, and a watery fluid accumulated. Desquamation of the respiratory and olfactory epithelia was evident. Alveolar capillaries were slightly congested, but edema and inflammatory cells were absent. Martin and Willoughby (1971) reported loss of cilia, disappearance of goblet cells, and metaplasia of the epithelium of the nasal cavity of pigs exposed

TABLE 12-1. LETHAL EFFECTS OF SO_2

SO_2 Concentration mg/m³	ppm	Duration	Species	Remarks	Reference
26.2	10	6 h/day x 5 day/wk x 113 day	Rat	No mortality in excess of control	Laskin et al., 1970
134	51	113 days	"	No mortality in excess of control	"
275	105	22 day	"	64% mortality (treated-control)	"
(See Text)		5 min/day x 5 day/wk > x lifetime	Mice	No increased mortality; tumor formation found	Peacock and Spence, 1967
1,598	610	LT_{50} 285.6 min	Mice (Connaught Med. Res. Lab. Strain)	IP injection of 200 to 300 mg histamine/mouse increased toxicity	Leong et al. 1961
2,392	913	74.5 min			
3,086	1,178	38.7 min	"	"	"
5,175	1,975	LT_{50} 197.6 min	Rat (Sprague-Dawley)	IP injection of 200 to 300 mg histamine/rat or adrenalectomy increased toxicity	Leong et al. 1961
9,165	3,498	71.7	"	"	"
13,236	5,052	41.0	"		"
5,782	2,207	LT_{50} 68.2 min	Guinea Pig	"	Leong et al. 1961
6,571	2,508	28.7	"		"
7,205	2,750	35.5	"		"
786	300	30 min	Guinea Pig	Increased mortality due to anaphylaxis from antigen challenge to sensitized animals	Matsumura, 1970a,b

TABLE 12-2. EFFECTS OF SO_2 ON LUNG MORPHOLOGY

Concentration	Duration	Species	Results	Reference
0.34, 2.65, or 15.0 mg/m^3 (0.13, 1.01, or 5.72 ppm) SO_2	1 yr., continuous	Guinea pig	Lungs of 15.0 mg/m^3 (5.72 ppm) group showed less spontaneous pulmonary disease than controls, and 0.34 and 2.64 mg/m^3 (0.13 and 1.01 ppm) animals. Tracheitis present in all but 15.0 mg/m^3 (5.72 ppm) group. Survival greater in the high dose group.	Alarie et al., 1970
0.37, 1.7 or 3.35 mg/m^3 (0.14, 0.64 or 1.28 ppm) SO_2	78 wk, continuous	Cynomolgus monkey	No remarkable morphologic alterations in the lung	Alarie et al., 1972, 1973c
12.3 mg/m^3 (4.69 ppm) then between 524 and 2620 mg/m^3 (200-1000 ppm) then 0 mg/m^3 SO_2	30 wk then 1 h then 48 wk	Cynomolgus monkey	Persistent changes in lung morphomology, including alterations in the respiratory bronchioles, alveolar ducts, and alveolar sacs.	Alarie et al., 1972, 1973c
13.4 mg/m^3 (5.12 ppm) SO_2	18 mo, continuous	Cynomolgus monkey	No alterations in lung morphology	Alarie et al., 1975
13.4 mg/m^3 (5.1 ppm) SO_2	21 h/day, 620 days	Dog	No alterations in lung morphology	Lewis et al., 1973
26.2 mg/m^3 (10 ppm) SO_2	72 h, continuous	Mouse	Pathological changes in the nasal mucosa. No alterations of tracheae; slight congestion of alveolar capillaries, but no alveolar edema (mice free of upper respiratory pathogens were significantly less affected than conventionally raised animals.)	Giddens & Fairchild, 1972
91.7 mg/m^3 (35 ppm) [rose on occasion to 262 mg/m^3 (100 ppm)] SO_2	1 to 6 wk	Pig	Loss of cilia in nasal cavity, disappearance of goblet cells, metaplasia of the epithelium	Martin & Willoughby, 1971
131, 262, 542, 786 mg/m^3 (50, 100, 200, 300 ppm) SO_2	3 h/day, 5 day/wk, 6 wk	Rat	Trachael goblet cells increased in number and size. Incorporation of $^{35}SO_4^{2}$ into mucus increased. Sialidase resistant mucus secreting cells were found much more distally. Chemical composition of mucus altered.	Reid, 1970
1048 mg/m^3 (400 ppm) SO_2	3 h/day, 5 day/wk, 3 wk	Rat	Increased mitosis of goblet cells. Alteration not lost by 5 wk postexposure.	Lamb and Reid, 1968

1 ppm SO_2 = 2.62 mg/m^3.

to 91.7 mg/m^3 (35 ppm) SO_2 for 1 to 6 weeks. This study, however, was marred by difficulties with the control of the SO_2 concentration, which rose on occasion to 262 mg/m^3 (100 ppm), and with high relative humidity (RH) occurring during cleaning of the pig pens.

Lamb and Reid (1968) and Reid (1970) attempted to use SO_2-exposed rats as a model of human chronic bronchitis. They presented arguments that SO_2-induced bronchial hyperplasia is analogous to human chronic bronchitis. Most of their studies were carried out at high concentrations of SO_2 (1,048 mg/m^3 or 400 ppm SO_2 for 3 h/day, 5 days/wk) for up to 6 weeks. Under these conditions, the tracheal goblet cells clearly increased in number and size. The goblet cell density also increased in the proximal airways, main bronchi, trachea, and distal airways, with proximal airways and main bronchi showing the largest changes. The incorporation of ^{35}S-sulfate into mucus by goblet cells also increased with exposure, reaching a plateau at approximately 3 weeks. The effects of SO_2 were concentrated in the central airways, again suggesting that the solubility of SO_2 in water limits its accessibility to the periphery. Mitosis reached a maximum after 2 or 3 exposures and declined rapidly as injured cells were replaced. On repeated exposure up to 6 wk, mitosis remained elevated in the proximal airways compared to the distal airway, in which the mitotic index returned to the control level. The magnitude was proportional to the SO_2 concentration up to 524 mg/m^3 (200 ppm) but was less at 786 mg/m^3 (300 ppm). The mitotic index rose at SO_2 concentrations as low as 131 mg/m^3 (50 ppm) when given for 3 h/day, 5 days/week. Major changes in the goblet cell type or substance produced by the goblet cells were also detected. Goblet cells, which produce mucus resistant to digestion by sialidase, increased in number, and their distribution extended distally from the upper bronchioles towards the respiratory bronchioles. Since each molecular type of mucin, sialidase-resistant or -susceptible, could be produced by one type of goblet cell, or each goblet cell could produce different mucins, these results can be interpreted in two ways. The elaboration of a specific type of goblet cell could occur, or more goblet cells could be produced but with a change in their biochemical function towards sialidase-resistant mucins.

Goldring et al. (1970) observed similar changes in morphology in Syrian hamsters exposed to 1700 mg/m^3 (650 ppm) SO_2 for 4 h/day over several weeks. These alterations included stimulation of mucus cell secretion and "dysplastic" changes in the bronchial epithelium. While these studies, along with those of Lamb and Reid, present an interesting means of studying experimental bronchitis, they do not provide evidence that ambient SO_2 levels cause similar changes.

Alarie et al. (1970) examined the tissues of guinea pigs exposed continuously to 0, 0.34, 2.65, or 15.0 mg/m^3 (0, 0.13, 1.01, or 5.72 ppm) SO_2 for 1 year. The lungs of the guinea pigs exposed to 15.0 mg/m^3 (5.72 ppm) and killed after 13 or 52 wk of exposure showed less spontaneous pulmonary disease than the control group. The prevalence of pulmonary disease in the control groups, which was not observed prior to exposure, suggests that they acquired pulmonary disease during the exposure period. This and other studies by Alarie and coworkers (1972, 1973c, 1975), were limited to light microscopic observations of conventional hematoxylin-eosin stained paraffin sections. The results are of limited value compared to more recent approaches using scanning electron microscopy of surfaces, transmission electron microscopy of organelles, or morphometric techniques. The control group, as well as those exposed to 0.34 and 2.64 mg/m^3 (0.13 and 1.01 ppm) SO_2, had evidence of lung disease as shown by histocytic infiltration of the alveolar walls. Tracheitis was also present in these three groups, but not in the 15.0 mg/m^3 (5.72 ppm) group. The latter group developed hepatocyte vacuolation, but the pathological significance of this change needs further investigation. The authors did not address this issue. The survival was greater ($p < 0.05$) in the 15.0 mg/m^3 (5.72 ppm) group than in the other groups, including the air-exposed control group. The possible effects of SO_2 in the Alarie et al. (1970) study, however, cannot be determined accurately because of the disease in the control animals.

Alarie et al. (1972, 1973c) subsequently exposed cynomolgus monkeys continuously to 0.37, 1.7 or 3.35 mg/m^3 (0.14, 0.64 or 1.28 ppm) SO_2 for 78 wk but found no remarkable morphological alterations. Another group exposed to 12.3 mg/m^3 (4.69 ppm) SO_2 for 30 wk was accidentally exposed to concentrations of SO_2 not higher than 2,620 mg/m^3 (1,000 ppm) or lower than 524 mg/m^3 (200 ppm) for 1 h, after which they were placed in a clean air chamber and held for 48 more weeks. Persistent changes were noted in this group. Alterations in the respiratory bronchioles, alveolar ducts, and alveolar sacs were found. Proteinaceous material was found within the alveoli. The distribution of such lesions was focal, but was observed within all lobes of the lung. Alveoli containing proteinaceous material were generally those that arose directly from respiratory bronchioles. Alveolar walls were thicker and infiltrated with histocytes and leukocytes. Macrophages were present within these foci. Moderate hyperplasia of the epithelia of the respiratory bronchioles was found, and frequently the lumina of the respiratory bronchioles were plugged with proteinaceous material, macrophages, and leukocytes. Bronchiectasis and bronchiolectasis were present in 8 of 9 monkeys. Vacuolation of hepatocytes was also observed, as with the guinea pig group exposed to 15.0 mg/m^3 (5.72 ppm) SO_2 in the prior Alarie et al. (1970) study.

In a replication of this study, cynomolgus monkeys were exposed to 13.4 mg/m^3 (5.12 ppm) SO_2 continuously for 18 mo (Alarie et al., 1975). No alterations in lung morphology were reported to be due to SO_2. The morphological alterations reported in the control group included lung mite infections and associated "slight subacute bronchiolitis, alveolitis, and bronchitis." Pulmonary function measurements were made in the above mentioned studies (Alarie et al., 1970, 1972, 1973c, 1975) and are described in Section 12.2.4.

The absence of SO_2-induced morphological alterations as reported by Alarie et al. (1970, 1972, 1973c, 1975) and Lewis et al. (1973), who exposed dogs for 620 days (21 h/day) to 13.4 mg/m^3 (5.1 ppm) SO_2, is not unexpected considering the transient bronchoconstriction induced by acute SO_2 exposure reported by Amdur (1973) at lower concentrations (see Section 12.2.4). Alarie et al. (1970) pointed out, "As recent literature attests, there is also an obvious lack of knowledge about the correlation between subtle microscopic alterations in the lung and concomitant changes in this physiological parameter (lung function)." Further, the transient nature of the pulmonary function effects observed during short-term exposures would be difficult to detect morphologically unless the lungs were fixed during the time of exposure. Even then, if the cause of the increased pulmonary resistance were a subtle alteration of smooth muscle tone, as has been hypothesized, it might be morphologically undetectable.

Most of the studies in which the lungs of SO_2-exposed animals have been examined center around tracheitis, bronchitis, ulceration, and mucosal hyperplasia (Table 12-2; see also Reid, 1970). The lowest concentrations of SO_2 at which these alterations have been reported have been in the rat at 131 mg/m^3 (50 ppm) for 30 to 113 days (Reid, 1970). At higher concentrations (1,048 mg/m^3 or 400 ppm SO_2 for 3 h/day, 5 days/wk for 3 wk), recovery to normal morphology did not occur after 5 wk postexposure (Reid, 1970). The possibility of recovery from lower concentrations and shorter durations of exposure .is not known (Lamb and Reid, 1968; Reid, 1970). Discounting their first study (Alarie, et al., 1970), where the control group of guinea pigs had a higher level of pulmonary infection than the exposed groups, Alarie et al. (1973c) reported no effect from SO_2 exposure up to 5 ppm (13.1 mg/m^3). These observations were, however, restricted to light microscopy and did not include scanning or transmission electron microscopic observations. This group also reported no observable effects at 0.37, 1.7 or 3.35 mg/m^3 (0.14, 0.64, and 1.28 ppm) SO_2 (Alarie et al., 1975). The group of monkeys exposed to 12.3 mg/m^3 (4.69 ppm) is difficult to evaluate due to the accidental exposure to high levels of SO_2 (Alarie et al., 1972). No effects were reported for monkeys exposed to 13.5 or 13.7 mg/m^3 (5.15 or 5.23 ppm) SO_2 and H_2SO_4 aerosols at 0.10 mg/m^3 or fly ash at 0.44 mg/m^3 (Alarie et al., 1975).

Because of the transient bronchoconstrictive effects of SO_2, conventional light microscopic morphological studies are not likely to be useful in evaluating the acute effects of SO_2 exposure. Persistent alterations have not been noted at less than 131 mg/m^3 (50 ppm).

12.2.4 Alterations in Pulmonary Function

Changes in breathing mechanics have been among the most sensitive parameters of SO_2 toxicity. They have also been useful in studying the effects of aerosols alone or in combination with SO_2 (Sections 12.3.3.1 and 12.4.1.1). A variety of methods have been used, some of which have been applied to human exposures. A method for measuring increases in flow resistance due to bronchoconstriction in guinea pigs has been developed by Amdur (Amdur and Mead, 1955, 1958). Animals are not anesthetized and breathe spontaneously, allowing sensitive measurements of pulmonary function. Another method by Alarie and coworkers (1973d) measures changes in respiratory rate.

Several investigators (Nadel et al. 1965a; Corn et al., 1972; Frank and Speizer, 1965; Balchum et al., 1960; Nadel et al., 1965b) found that bronchoconstriction resulted from both head-only and lung-only exposures in cats and dogs. When corrected for the amount of SO_2 hypothesized to reach the lung, Amdur's study (1966) with guinea pigs showed that SO_2 is highly effective in producing bronchoconstriction through direct exposure of the lung. Two sets of receptors are involved in the response of animals to SO_2. At high concentrations of SO_2 or following lengthy exposure, the nasopharyngeal receptors fatigue or become unresponsive, whereas the bronchial receptors do not (Alarie, 1973). Widdicombe (1954a) originally described the receptors responsible for the SO_2-initiated bronchoconstriction. The bronchoconstriction is initiated through the activation of bronchial epithelial chemoreceptors whose efferent and afferent pathways are through the vagus nerves (Nadel et al., 1965a,b; Grunstein et al., 1977; Tomori and Widdicombe, 1969). Chilling the vagus prevents conduction of nervous impulses produced on inhalation of SO_2. Other receptors located in the same regions of the lung respond to mechanical stimulation and particles such as talc (Widdicombe, 1954b; Widdicombe et al., 1962). Intravenous injection of atropine blocks the efferent impulses, presumably at the cholinergic preganglionic synapse (Grunstein et al., 1977). Sulfur dioxide-initiated bronchoconstriction involves smooth muscle contraction, since β-adrenergic agonists such as isoproterenol reverse the SO_2-bronchoconstriction (Nadel et al., 1965a,b). Histamine may be involved in this response, as implied by other studies of hyperreactive airways (Boushey, et al. 1980), but definitive proof of histamine involvement is not available. Release of acetylcholine could also cause increased mucus secretion as noted during SO_2 exposure. Chronic exposure to SO_2 could lead to mucus hypersecretion and altered airway caliber. Cholinomimetic drugs and histamine applied as aerosols mimic the SO_2-initiated bronchoconstriction (Islam et al., 1972). Using anesthetized, intubated, spontaneously breathing dogs exposed to 2.62, 5.24, 13.1, or 26.2 mg/m^3 (1, 2, 5, or 10 ppm) SO_2 for 1 h, Islam et al. (1972) found an increased bronchial reactivity to aerosols of acetylcholine, a potent bronchoconstrictive agent. The greatest response occurred at 5.24 mg/m^3 (2 ppm), although 2.62 mg/m^3 (1 ppm) also caused an effect. The effect at 26.2 mg/m^3 (10 ppm) was less than that at 2.62, 5.24, and 13.1 mg/m^3 (1, 2, and 5 ppm). While these results could suggest that SO_2 may modify bronchial reactivity, the authors point out that the reactions were highly reversible and occurred in ranges where alterations can also be produced by the inhalation of saline aerosols. Cholinomimetic drugs act through either the same autonomic reflex arc or directly upon the cholinergic receptors on smooth muscles and mucus secreting cells and glands. As discussed in Chapter 13, SO_2 also produces bronchoconstriction in man through the same autonomic reflex arc.

Exposure to SO_2 increases resistance to air flow in guinea pigs that can be repeated by numerous exposures over several hours and exhibits none of the tachyphylaxis found with other species (Corn et al., 1972; Frank and Speizer, 1965). However, different techniques were used for different species. In a review of her data, Amdur (1973), reported that for a 1-h exposure, a mean of 0.68 mg/m^3 or 0.26 ppm (range of 0.08 to 1.57 mg/m^3 or 0.03 to 0.6 ppm) was the lowest concentration of SO_2 that increased flow resistance in guinea pigs. The response, a 12.8 percent increase ($p < .001$) at these low levels of SO_2 (Amdur, 1973), was the average for 71 guinea pigs; the individual data points were reported in other publications (Amdur and Underhill, 1968, 1970; Amdur, 1974). For a 1-h exposure, the lowest concentration these researchers tested that caused an increase ($p < 0.01$) in resistance was 0.42 mg/m^3 (0.16 ppm) SO_2 (Amdur and Underhill, 1970). In a more recent study, Amdur et al. (1978) showed that a 1-h exposure of guinea pigs to 0.84 mg/m^3 (0.32 ppm) SO_2 caused a 12 percent increase in resistance ($p < 0.02$) and a nonstatistically significant decrease in compliance. Investigations of the interaction of oil mists and SO_2 showed that 2.62 mg/m^3 (1 ppm) SO_2, the lowest concentration used, significantly increased resistance (Costa and Amdur, 1979a,b). At concentrations of SO_2 below 2.62 mg/m^3 (1 ppm), the response of individual animals varied considerably (Amdur, 1964, 1973, 1974). Of 1,028 guinea pigs, 135 were "susceptible," responding to low concentrations of SO_2 (and a variety of irritants) with greater changes in resistance than the predicted mean. Amdur cites comparative data for other species, including man, to suggest that a certain fraction of all subjects may exhibit this phenomenon (Amdur, 1973, 1974; Horvath and Folinsbee, 1977). It might also be suggested that some groups of animals by chance may not have a "susceptible" individual. In one study (Amdur et al., 1978), 3 groups of 10 animals each exposed to 0.52, 1.05, or 2.1 mg/m^3 (0.2, 0.4, or 0.8 ppm) SO_2 had no significant increase in airway resistance above the control values. It must be noted, however, that these results might also reflect intraspecies differences in susceptibility.

Based on data from earlier work (Amdur and Underhill, 1968), Amdur concluded that 10 to 13 percent of the guinea pig population is very much more responsive than the average (Amdur, 1974). Cats (Corn et al., 1972) and dogs (Frank and Speizer, 1965), on the other hand, rarely were found to be sensitive to short-term (< 1 h) exposure to 52.4 mg/m^3 (20 ppm) SO_2 (cats) or 18.3 mg/m^3 (7 ppm) SO_2 (dogs). Even with the relatively small sample sizes used, some cats and dogs responded and others did not.

Some of the problem of "susceptible" vs. "nonsusceptible" members of the experimental populations can be understood if one assumes that the response to a given toxicant, such as SO_2, is the result of a number of different genes within the population and not just a single gene. In that case, a single individual could have a number of recessive or dominant genes that could contribute to either "susceptibility" or "nonsusceptibility." Since experimental animals and human subjects are selected as randomly as possible (in most experimental designs), there is a maximal chance of getting some "susceptible" responders in each experiment. The total number of "susceptible" responders will be small and variable because of the low incidence of "susceptible" responders in the general animal population, but will tend to shift the dose- or concentration-response curve toward lower concentrations and to decrease the slope of the curve (e.g., when the data are expressed as the log-probit transformation). Such phenomena have been studied in detail for "resistant" insects that have different genomes responsible for increased detoxification mechanisms. In the case of SO_2, the matter is further complicated by comparisons between groups of animals and different strains or species. Even with guinea pigs, the total number of animals examined to date (about 1,000 to 2,000) is too small to give more than a crude estimate of those animals that have a "sensitive" genome. The incidence of "susceptibility" in the guinea pigs (about 13 percent) is too low to have been detected clearly in the 100 or so cats and dogs used in SO_2 experiments. Here only 1 or 2 "susceptible" animals would have been encountered in each experiment. Further, the small number of animals has been studied in different laboratories and at different times, and the animals have come from different genetic stocks. It is fortuitous that Amdur's laboratory has persisted in these studies with the same animal, the guinea pig, and the same general method, so this low incidence of "susceptibility" could be detected. While the mechanism(s) responsible for "susceptibility" is not known, the question of "susceptibility" is an important aspect deserving further study. A similar incidence of "susceptible" individuals found to exist in man would present a major health problem. Adverse reactions might conceivably occur among these individuals at exposures less than 2.62 mg/m^3 (1 ppm) (Amdur, 1964, 1973, 1974), which are encountered in ambient air. However, the frequency of susceptibility to SO_2 in man, as well as the physiological or biochemical basis of such susceptibility, is not known.

A broad dose-response curve has been noted also for histamine-initiated bronchoconstriction in man (Habib et al., 1979), guinea pigs (Douglas et al., 1973, 1977; Brink et al., 1980), dogs (Loring et al., 1978; Snapper et al., 1978), and monkeys (Michoud, 1978). Among 12 normal human subjects, a 38-fold range of inhaled histamine was observed in both the threshold and median doses causing bronchoconstriction (Habib et al., 1979). The concentration required to produce a 50-percent change in dynamic lung compliance in 131 female guinea pigs varied over a 100-fold range (Douglas et al., 1973). While the interindividual amount varied considerably, the values were lognormally distributed, indicating a single population (Douglas et al., 1973, 1977). Dogs showed a 40-fold variation in histamine concentration needed to initiate changes in airway diameter (Snapper et al., 1978). These values were also log-normally distributed, indicating a single population among the 102 mongrel dogs examined. A wide interindividual variation for histamine- and methacholine-initiated bronchoconstriction was found among 8 rhesus monkeys, some of which were sensitive to Ascaris suum allergen (Michoud, 1978). The sensitivity to histamine or methacholine was not associated with Ascaris sensitivity, however. While genetic differences in histamine sensitivity have been found in guinea pigs, naturally occurring or acquired allergic reactions are not likely to cause the large interindividual differences in sensitivity in either guinea pigs (Takino et al., 1971) or monkeys (Michoud et al., 1978). A further complicating factor is the age-dependence of histamine- and other drug-initiated bronchoconstriction (Brink et al., 1980). Younger guinea pigs are more sensitive to histamine than older animals. This decreasing bronchial reactivity to histamine with age in the guinea pig has been suggested as a model of human juvenile asthma.

Human bronchial hyperreactivity does not seem to decrease with age in the same manner, however (Boushey et al., 1980). While large interindividual differences apparently occur with a wide variety of chemical agents causing bronchial reactivity in both man and animals, the response of the same individual is quite reproducible regardless of species. The variability in the lowest dose of SO_2 needed to evoke a given bronchoconstriction (measured, for example, as an increased resistance to flow by the studies of Amdur) is apparently an inherent part of the bronchial response to a broad range of chemicals and is not an artifact of the method. Similar variations in lowest effective doses for SO_2 are likely to occur in man, judging from the variability of response to inhaled histamine. The general observation that asthmatic patients appear to be hypersensitive to a broad range of chemical and physical agents initiating bronchoconstriction (Boushey et al., 1980; see also Sect. 13.2.3) supports the contention that the most susceptible animal species might possibly be used experimentally as a surrogate for man. A major difference in pharmacology may exist between the guinea pig and man, however. For example, autonomic mediators interact with histamine-induced bronchial reactivity in guinea pigs but not in man, and beta adrenergic blockade by propranolol causes no difference in bronchial reactivity in man (Habib et al., 1979) but potentiates histamine reactivity in the guinea pig (Douglas et al., 1973). Insufficient numbers of animals and subjects have been examined to predict the general shape of the dose-response curve for the human population, even excluding the hypersensitive asthmatic population. These variations in interindividual dose needed to evoke a specific amount of increased resistance to flow in guinea pigs by SO_2 likewise apply to the measurement of increased resistance to flow evoked by aerosols, as discussed below in Section 12.3.3.

Using Strandberg's (1964) data from the rabbit to correct for the concentration of SO_2 hypothesized to reach the lung, Amdur (1966) was able to normalize the concentration-response curve for SO_2-induced bronchoconstriction in the guinea pig resulting from nose-only exposures. A break occurs in the concentration-response curve at about 52.4 mg/m^3 (20 ppm) SO_2, perhaps due to the poorer extraction of gaseous SO_2 by the upper airways at low concentrations. It should be recognized, however, that SO_2-extraction data for rabbits (Strandberg, 1964) and dogs (Frank et al., 1967; Balchum et al., 1960; Frank et al., 1969) are in some conflict and that the data for rabbits are not clear with respect to the site of SO_2 removal. Thus, use of the rabbit data for guinea pig studies can be done only hypothetically. Sulfur dioxide introduced directly into the lung by a tracheal cannula was much more effective in producing bronchial constriction. Amdur (1966) suggested that, at concentrations of 1.05 to 1.31 mg/m^3 (0.4 to 0.5 ppm), very little removal of SO_2 occurs in the upper airways. These data contrast with the radiotracer studies in dogs (Frank et al., 1967, 1969; Balchum et al., 1960). Others have required concentrations greater than 18.3 mg/m^3 (7 ppm) to evoke increases in flow resistance in anesthetized cats (Corn et al., 1972) and dogs (Frank and Speizer, 1965). Differences in the sensitivity of the two models may lie in the use of anesthesia, in the use of different species, or in a different incidence of "susceptible" individuals.

Animals chronically exposed to SO_2 have also been examined for alterations in pulmonary function. Guinea pigs exposed continuously to 0.34, 2.64, or 15 mg/m^3 (0.13, 1.01, or 5.72 ppm) SO_2 for up to 1 yr showed no changes in pulmonary function; however, spontaneous pulmonary disease was present in all animals (including controls) except those exposed to the highest concentration (Alarie et al., 1970). Dogs exposed for 21 h/day to 13.4 mg/m^3 (5.1 ppm) SO_2 for up to 225 days demonstrated increased pulmonary flow resistance and decreased lung compliance (Lewis et al., 1969). After exposure for 620 days, the mean nitrogen washouts of dogs increased (Lewis et al., 1973). Alarie and coworkers (Alarie et al., 1972, 1973c, 1975) exposed cynomologus monkeys continuously to 0.37, 1.7, 3.4, or 13.4 mg/m^3 (0.14, 0.64, 1.28, or 5.12 ppm) SO_2. The latter concentration was used in an 18-mo study, whereas the others were used for 78-wk exposures. Pulmonary function remained unchanged in all of these groups. After 30 wk of exposure to 12.3 mg/m^3 (4.69 ppm) SO_2, the monkeys were inadvertently exposed to concentrations between 524 and 2,620 mg/m^3 (200 and 1000 ppm) for 1 hour. This treatment resulted in pulmonary function alterations that persisted for the remaining 48 wk of the study, during which the animals were exposed to clean air. Morphological alterations were also seen in this group (see Section 12.2.3).

The respiratory rate of mice has been used as an indication of sensory irritation by Alarie et al. (1973d). Mice were exposed for 10 min to 0, 44.5, 83.8, 162, 233, 322, 519, or 781 mg/m^3 (0, 17, 32, 62, 89, 123, 198, or 298 ppm) SO_2. About a 12 percent decrease in respiratory rate was observed at 44.5 mg/m^3 (17 ppm). The respiratory rate decreased inversely to the logarithm of the concentration of inspired SO_2. The decrease in respiratory rate was transient, however, as complete recovery to control values occurred within 30 min following all exposures to SO_2. The time for maximum response was inversely related to the logarithm of the concentration of SO_2, being shortest at highest concentrations. Mice exposed to 262 mg/m^3 (100 ppm) SO_2 for 10 min were allowed to recover in clean air prior to a subsequent 10 min exposure to the same concentration. As the length of the recovery period was decreased (from 12 min to 3 min), the effect of the subsequent SO_2 exposure on respiratory rate was lessened. "Desensitization" thus appeared to occur during the course of exposures. When another irritant, aerosols of chlorobenzilidene malononitrile (CBM), was used during the refractory period following SO_2 exposure, the respiratory rate decreased at a rate comparable to that following exposure to CBM alone. Thus, the refractory period associated with SO_2 exposures appeared specific to SO_2 and not to CBM. When 262 to 328 mg/m^3 (100 to 125 ppm) SO_2 was provided repeatedly for 90 sec, with each exposure separated by a 60-sec recovery period, the refractory period was cumulative. Ten such exposures eventually abolished all respiratory rate responses to SO_2. Breathing clean air for 60 min resulted in a return of the response to initial levels. When mice were exposed to SO_2 by means of a tracheal cannula, no changes in the respiratory rate were observed, indicating that the decrease in respiratory rate was mediated by a reflex arc. This concept has been developed in considerable detail in an extensive review by Alarie (1973), who suggests that stimulation and desensitization occur via cholinergic nerve endings of the afferent trigeminal and glossopharyngeal nerves, allowing activation of receptors in the nose and upper airways. Alarie et al. (1973d) also suggest that SO_2 is hydrated to bisulfite and sulfite that react with a receptor protein to form an S-thiosulfate and a thiol, cleaving an existing disulfide bond. The receptor protein slowly regenerates to its original disulfide configuration by the oxidation of S-thiosulfide and free thiol moieties of the receptor protein to disulfide. No direct evidence for this hypothesis has been presented, however.

In summary, decreases in respiratory rate or increased resistance to flow are reproducible end points. There are at least two sets of receptors responsible for these changes in respiratory function in animals acutely exposed to SO_2. Increased resistance to flow results from SO_2 concentrations as low as 0.42 mg/m^3 (0.16 ppm) in guinea pigs. Of the animals so far examined, guinea pigs are the most sensitive to SO_2. The reason for this is not known, but potential factors include species, strains, and experimental technique used. Large interindividual differences in dose-response curves for changes in pulmonary resistance to airflow exist in all species. The exact number of animals responding to a given dose depends on the shape of the dose-response curve. The nature of the dose-response curve at low levels is poorly understood and has not been investigated directly. While pulmonary function in guinea pigs appears to be highly sensitive to acute SO_2 exposures, it has not been proven that chronic SO_2 exposures have a similar effect. Chronic studies with guinea pigs are unclear because of disease in the control group. In other chronic studies, pulmonary function of monkeys was unchanged at SO_2 concentrations up to 13.4 mg/m^3 (5.12 ppm); dogs were affected by 225, but not 620, days of exposure to 13.4 mg/m^3 (5.1 ppm). High levels of SO_2 likely to initiate airway narrowing and hypersecretion of mucus do alter several parameters of pulmonary function. These results are not contradictory in view of the physiology of SO_2-initiated bronchoconstriction. Sulfur dioxide appears to cause bronchoconstriction through action on the smooth muscles surrounding the airways. Since smooth muscles fatigue or become adjusted to altered tone over time, chronic exposure to SO_2 is not likely to cause a permanent alteration in bronchial tone. Unfortunately, investigations of the reactions of the airways after chronic exposure to SO_2 have not appeared. We do not know if chronic exposure to SO_2 causes an alteration in response to SO_2 itself, since only direct measurements of pulmonary function were made on the animals after chronic exposure. It would be informative to learn if chronically-exposed monkeys, for example, were more or less sensitive to SO_2 (Table 12-3).

12.2.5 Effects on Host Defenses

Because alterations in the ability to remove particles from the lung could lead to increased susceptibility to airborne microorganisms or increased residence times of other nonviable particles, the effects of SO_2 on particle removal and engulfment, as well as on integrated defenses against respiratory infection, have been studied. Cilia function does not appear to be affected by exposure. No changes were observed in the cilia beat frequency or

TABLE 12-3. EFFECTS OF SO_2 ON PULMONARY FUNCTION

Concentration	Duration	Species	Results	Reference
0.37, 1.7, 3.4, or 13.4 mg/m³ (0.14, 0.64, 1.28, or 5.12 ppm) SO_2	72-78 wk, continuous	Cynomologus monkey	No change	Alarie et al., 1972, 1973c, 1975
0.42 or 0.84 mg/m³ (0.16 or 0.32 ppm) SO_2	1 h	Guinea pig	Increase in airway resistance	Amdur et al., 1970, 1978a
0.52, 1.04, or 2.1 mg/m³ (0.2, 0.4, or 0.8 ppm) SO_2	1 h	Guinea pig	No significant increase in airway resistance	Amdur et al., 1978c
2.62, 5.24, 13.1, or 26.2 mg/m³ (1, 2, 5, or 10 ppm) SO_2	1 h	Dog	Increased bronchial reactivity to aerosols of acetylcholine, a potent bronchoconstrictive agent	Islam et al., 1972
13.4 mg/m³ (5.1 ppm) SO_2	21 h/day, 225 and 620 days	Dog	Increased pulmonary flow resistance and decreased lung compliance at 225 days; increased nitrogen washout at 620 days	Lewis et al., 1969,1973
0, 44.5, 83.8, 162, 233, 322, 519, or 781 mg/m³ (0, 17, 32, 62, 89, 123, 198, or 298 ppm) SO_2	10 min	Mouse	Decreased respiratory rate proportional to the log of the concentration; complete recovery within 30 min. The time for maximum response was inversely related to the log of the concentration.	Alarie et al., 1973d

1 ppm SO_2 = 2.62 mg/m³.

the relative number of alveolar macrophages laden with particles in rats exposed to 2.62 or 7.86 mg/m^3 (1 or 3 ppm) SO_2 and graphite dust (mean diameter 1.5 μm, 1 mg/m^3) for up to 119 consecutive days (Fraser et al., 1968). Donkeys (Spiegelman et al., 1968) were exposed by nasal catheters to 68.1 to 1,868 mg/m^3 (26 to 713 ppm) SO_2 for 30 min. Clearance was not affected below 786 mg/m^3 (300 ppm), but at high concentrations (786 to 1,868 mg/m^3 or 376 to 713 ppm), clearance was depressed.

Ferin and Leach (1973) exposed rats to 0.26, 2.62, and 52.4 mg/m^3 (0.1, 1, or 20 ppm) SO_2 for 7 h/day, 5 days/wk, for a total of 10 to 15 days and then measured the clearance of an aerosol of titanium oxide (TiO_2). The aerosol was generated at about 15 mg/m^3 (1.5 μm MMAD, σ_g=3.3). These investigators took the amount of TiO_2 retained at 10 to 25 days as a measure of the "integrated alveolar clearance." Low concentrations of SO_2 (0.26 mg/m^3 or 0.1 ppm) accelerated clearance after 10 and 23 days, as did 2.62 mg/m^3 (1 ppm) at 10 days. By 25 days, however, clearance was decreased with 1 ppm. Hirsch et al. (1975) found that the tracheal mucus flow was reduced in beagles exposed for 1 yr to 2.62 mg/m^3 (1 ppm) SO_2 for 1.5 h/day, 5 days/week. No differences in pulmonary function were reported. Confirmation of this study and determination of the persistence of the decreased mucus flow at this low level of SO_2 would be important in light of other data available.

Sulfur dioxide may have more of an effect on antiviral than on antibacterial defense mechanisms. Bacterial clearance was not depressed or altered in guinea pigs exposed to 13.1 or 26.2 mg/m^3 (5 or 10 ppm) SO_2 for 6 h/day for 20 days (Rylander, 1969; Rylander et al., 1970). Using the infectivity model (see Section 12.3.4.3), Ehrlich et al. (1978) found that short (3 h/ day for 1 to 15 days) or long (24 h/day for 1 to 3 mo) exposures to 13.1 mg/m^3 (5 ppm) SO_2 did not increase mortality subsequent to a pulmonary streptococcal infection. Virus infections, however, are augmented by simultaneous or subsequent SO_2 exposure. Mice were exposed to concentrations varying from 0 to 52.4 mg/m^3 (0 to 20 ppm) SO_2 continuously for 7 days (Fairchild et al., 1972). Mice breathing 18.3 to 26.2 mg/m^3 (7 to 10 ppm) SO_2 had an increase in pneumonia. Lung consolidation was significant at 65.5 mg/m^3 (25 ppm), but not at 26.2 or 39.3 mg/m^3 (10 or 15 ppm). The rate of growth of the virus within the lung was unaffected by SO_2 exposure. Further analysis of the data (Lebowitz and Fairchild, 1973) indicated that SO_2 and virus exposure produced weight loss at concentrations as low as 9.43 mg/m^3 (3.6 ppm). Exposure to SO_2, whether alone or in combination with a viral agent, had more of an effect on weight reduction than on pneumonia. Since Giddens and Fairchild (1972) showed that mice with apparent respiratory infection were more susceptible to SO_2 (Section 12.2.3), a rebound effect may be possible in which SO_2 and microbial agents each potentiate the effect of the other.

Several studies of the effects of SO_2 on alveolar macrophages have been conducted, since these cells participate in clearance of viable and nonviable particles in the gaseous exchange regions of the lung. Rats were exposed for 24 h to 2.62, 13.1, 26.2, and 52.4 mg/m^3 (1, 5, 10, and 20 ppm) SO_2 and their alveolar macrophages investigated by Katz and Laskin (1976).

Exposure to the two highest concentrations increased in vitro phagocytosis of latex spheres for up to 4 days in culture. At 13.1 mg/m^3 (5 ppm) SO_2, phagocytosis was increased after 3 or 4 days in culture, but not after 1 or 2 days. Histochemical studies of pulmonary macrophages from rats which had been exposed to 786 mg/m^3 (300 ppm) SO_2 for 6 h/day on 10 consecutive days showed no changes in the lysosomal enzymes, β-glucuronidase, β-galactosidase, and N-acetyl-β-glucosaminidase (Barry and Mawdesley-Thomas, 1970). Acid phosphatase activity was markedly increased. This is in agreement with Rylander's observation (Rylander, 1969) that suggests that SO_2 exposure (26.2 mg/m^3 (10 ppm) for 6 h/day, 5 days/wk for 4 wk) does not affect the bactericidal activity of the lung (see Table 12-4).

12.3 EFFECTS OF PARTICULATE MATTER

Sulfur dioxide is oxidized to sulfuric acid (H_2SO_4) in the atmosphere. Sulfuric acid can react with atmospheric ammonia (NH_3) to produce ammonium sulfate and bisulfate. Similar reactions can also occur in the animal exposure chamber and confound experiments. Ambient PM usually contains some proportion of sulfur compounds and the definition of the effects of ambient aerosols independent of sulfur compounds may be impossible. Sulfur dioxide is often present in polluted atmospheres with complex mixtures of other compounds including heavy metals, which may be present as oxides or as sulfate or nitrate salts. In addition, organic compounds present in the atmosphere in the gaseous phase can be associated with the particulate fraction or become adsorbed on particles either in situ or during collection. Inhalation of particles with surface coatings of toxic elements, organic compounds, allergens or gases (LaBelle et al., 1955) may result in greater effects due to localized surface reaction with lung tissue or macrophages (Camner et al., 1974). The diversity of these types of particles precludes discussion of their toxicity at this time, since little or no inhalation data are available. The details of the composition of atmospheric aerosols are covered in Chapter 2 and the deposition and transport of particles are discussed in Chapter 11.

Since very few studies have appeared on the toxicity of complex atmospheric particles themselves, this section primarily covers the toxicology of those compounds which have been identified as constituents of atmospheric particles. Therefore, these discussions, no matter how sophisticated for a single component, are inherently simplistic. For aerosols other than H_2SO_4, $(NH_4)_2SO_4$, and NH_4HSO_4, this information is integrated in the perspective of the potential biological effects of atmospheric particles.

As will be apparent from the discussion of the toxicity of sulfate aerosols in this section, the chemical composition of its constituent particles determines the toxicity of an atmospheric aerosol. Those particles that are biologically active may have direct toxic effects in themselves, indirect toxic effects through interactions with other pollutants, and chronic effects through cell transformation or chronic alteration in cell function. Direct

TABLE 12-4. EFFECTS OF SO_2 ON HOST DEFENSES

Concentration	Duration	Species	Results	Reference
0.26, 2.62, or 52.4 mg/m^3 (0.1, 1, or 20 ppm) SO_2	7 h/day, 5 day wk	Rat	Low concentrations (0.26 mg/m^3 or 0.1 ppm) accelerated alveolar clearance after 10 and 23 days, as did 2.62 mg/m^3 (1 ppm) at 10 days; at 25 days, 1 ppm decreased clearance.	Ferin and Leach, 1973
2.62, 13.1, 26.2, and 52.4 mg/m^3 (1, 5, 10, and 20 ppm) SO_2	24 h	Rat	Exposure to the two higher concentrations increased in vitro phagocytosis of latex spheres for up to 4 days in culture. At 13.1 mg/m^3 (5 ppm), phagocytosis was increased after 3 or 4 days in culture, but not 1 or 2 days.	Katz and Laskin, 1976
2.62 mg/m^3 (1 ppm) SO_2	1.5 h/day, 5 day/wk	Dog	Tracheal mucous flow was reduced.	Hirsch et al., 1975
2.62 or 7.86 mg/m^3 (1 or 3 ppm) SO_2 + graphite dust (mean diameter 1.5 μm, 1 mg/m^3)	Up to 119 days	Rat	No changes in the cilia beat frequency or the relative number of alveolar macrophages laden with particles.	Fraser et al., 1968
9.43 to 52.4 mg/m^3 (3.6 to 20 ppm) SO_2	7 days continuous	Mouse	Exposure to SO_2 and a virus produced weight loss	Lebowitz and Fairchild, 1973
13.1 or 26.2 mg/m^3 (5 or 10 ppm) SO_2	6 h/day, 20 day	Guinea pig	Bacterial clearance was not altered	Rylander, 1969, 1970
13.1 mg/m^3 (5 ppm) SO_2	3 h/day, 1-15 days and 24 h/day, 1-3 mo	Mouse	Did not increase mortality subsequent to a pulmonary streptococcal infection	Ehrlich, 1979
Varying from 0 to 52.4 mg/m^3 (0 to 20 ppm) SO_2	7 days, continuous	Mouse	Increase in viral pneumonia at 18.3 to 26.2 mg/m^3 (7 to 10 ppm). Rate of growth of virus unaffected.	Fairchild et al., 1972
26.2 mg/m^3 (10 ppm) SO_2	6 h/day for 20 days	Rat	Did not affect the bactericidal activity of the lung	Rylander, 1969
65.5 to 1868 mg/m^3 (25 to 713 ppm) SO_2	30 min	Donkey	Below 786 mg/m^3 (300 ppm), mucociliary clearance was not affected, but at high concentrations (786 to 1868 mg/m^3 or 376 to 713 ppm), clearance was depressed.	Spiegelman et al., 1968
786 mg/m^3 (300 ppm) SO_2	6 h/day, 10 days continuous	Rat	No changes in selected lysosomal enzymes	Barry et al., 1970

1 ppm SO_2 = 2.62 mg/m^3.

toxic effects are best substantiated by cytotoxicity studies. Those reviewed here are for some specific compounds that occur in the particulate fraction. The studies cited are by no means complete and could be expanded by including a number of other investigations carried out *in vitro* or by exposures other than inhalation. The review was purposefully restricted to those most applicable to the inhalation route of exposure. Major exceptions to this policy have been made for silica and the limited data on compounds in the so-called "coarse-mode" particles fraction. Most of the effects through interaction with other pollutants have previously been discussed for SO_2. Some additional data implicating interactions between SO_2 and PM or ozone, and between H_2SO_4 and ozone are included.

Almost all of the studies (and all of the inhalation studies) discussed in this section involve the health effects of particles in the "fine mode" size range and composition. Within this category, several investigators examined the influence of particle size for a given chemical. For coarse mode particles, only a few *in vitro* and intratracheal instillation studies could be found. This work is discussed separately.

12.3.1 Mortality

The susceptibility of laboratory animals to H_2SO_4 aerosols varies considerably. Amdur (1971) reviewed the toxicity of H_2SO_4 aerosols and pointed out that, of the commonly used experimental animals, guinea pigs are the most sensitive and most similar to man in their bronchoconstrictive response to H_2SO_4. The lethal concentration (LC) of H_2SO_4 depends on the age of the animal (18 mg/m^3 for 8 h for 1 to 2 mo-old vs. 50 mg/m^3 for 18 mo-old animals), the particle size (those near 2 µm being more toxic), and the temperature (extreme cold increasing toxicity). Wolff et al. (1979) found the LC_{50} (the concentration at which 50 percent of the animals die) in guinea pigs for an 0.8 µm (MMAD) aerosol to be 30 mg/m^3, whereas for a 0.4 µm (MMAD) aerosol it was above 109 mg/m^3. In determining acute toxicity, the concentration of the aerosol appears to be more important than the length of exposure (Amdur et al., 1952). The animals that died did so within 4 hours. Chronic studies have only recently been undertaken, and they support this conclusion that mortality rarely occurs at moderate concentrations of H_2SO_4.

Sulfuric acid aerosol appears to have two actions. Laryngeal and/or bronchial spasm are the predominant causes of death at high concentrations. When lower concentrations are used, bronchostenosis and laryngeal spasm can still occur. Pathological lesions in the latter case include capillary engorgement and hemorrhage. Such findings are in accord with anoxia as the primary cause of death.

12.3.2 Morphological Alterations

Alarie et al. (1973a) investigated the effects of chronic H_2SO_4 exposure. Guinea pigs were exposed continuously for 52 wk to 0.1 mg/m^3 H_2SO_4 (2.78 µm, MMD) or to 0.08 mg/m^3 H_2SO_4 (0.84 µm, MMD). Monkeys were exposed continuously for 78 wk to 4.79 mg/m^3 (0.73 µm, MMD), 2.43 mg/m^3 (3.6 µm, MMD), 0.48 mg/m^3 (0.54 µm, MMD), or 0.38 mg/m^3 H_2SO_4, (2.15 µm, MMD).

Sulfuric acid had no significant hematological effects in either species. No light microscopic lung alterations resulting from H_2SO_4 exposure were observed in guinea pigs after 12 or 52 wk of exposure in this study (Alarie et al., 1973a) or in a later study (Alarie et al., 1975). Morphological changes were evident in the lungs of monkeys. At the two highest concentrations, there were changes (more prevalent in the 4.79 mg/m^3 H_2SO_4 group) regardless of the particle size. Major findings included bronchiolar epithelial hyperplasia and thickening of the walls of the respiratory bronchioles. Alveolar walls were thickened in monkeys exposed to 2.43 but not to 4.79 mg/m^3 H_2SO_4. Particle size, however, had an impact at lower H_2SO_4 concentrations. No significant alterations were seen after exposure to 0.48 mg/m^3 of the smaller particle size (0.54 μm). However, bronchiolar epithelial hyperplasia and thickening of the walls of the respiratory bronchioles were seen after exposure to the larger size (1.15 μm) and lower concentration (0.38 mg/m^3). Pulmonary function changes followed a slightly different pattern (see Section 12.3.4.2). Dogs also appear to be relatively insensitive to H_2SO_4 alone, as judged by morphological changes. Lewis et al. (1973) found no morphological changes after the dogs had been exposed for 21 h/day for 620 days to 0.89 mg/m^3 H_2SO_4 aerosol (90 percent of the particles were <0.5 μm in diameter).

Cockrell and Busey (1978) and Ketels et al. (1977) studied the morphological changes resulting from H_2SO_4 aerosols. Cockrell and Busey (1978) examined the effects of 25 mg/m^3 H_2SO_4 (1 μm, MMD, σ_g 1.6) for 6 h/day for 2 days in guinea pigs. Segmented alveolar hemorrhage, type 1 pneumocyte hyperplasia, and proliferation of pulmonary macrophages were reported. Ketels et al. (1977) examined the response of mice to 100 mg/m^3 H_2SO_4; these exposures produced injury to the top and middle of the trachea, but none to the lower trachea and distal airways. In an investigation of the dose-response relationship for H_2SO_4, mice received either 5 daily 3 h-exposures to 200 mg/m^3, 10 daily exposures to 100 mg/m^3, 20 daily exposures to 50 mg/m^3, or any one of these doses combined with 5 mg/m^3 carbon particles. The damage was judged to be proportional to the concentration (C) of H_2SO_4, but not to the integrated dose (C x T) or to the time of exposure (T). (All of the exposures had the same C x T and therefore their equivalence might have been hypothesized.)

A number of other studies of the morphological effects of H_2SO_4 when combined with other pollutants have been conducted. (See Section 12.4.1.2. and Table 12-5)

Inhalation of silicon dioxide (SO_2) results in silicosis, which is characterized by morphological changes in the lungs. Because the extensive information on silicosis has been reviewed elsewhere (Ziskind et al., 1976; NIOSH, 1975; Reiser and Last, 1979; Singh, 1978), it is not discussed in detail here. Due to the toxicity of SiO_2, a Threshold Limit Value (American Conference of Governmental Hygenists, 1979) has been set. Because of the involvement of alveolar macrophages in its toxicity and its presence in ambient particles, however, some of the effects of SiO_2 are summarized briefly here. All the information given below for silicon is derived from reviews by Ziskind et al. (1976) and NIOSH (1975).

TABLE 12-5. EFFECTS OF H_2SO_4 AEROSOLS ON LUNG MORPHOLOGY

Concentration	Duration	Species	Results	Reference
0.08 mg/m³ H_2SO_4 (0.84 μm, MMD), or 0.1 mg/m³ H_2SO_4 (2.78 μm, MMD)	52 wk, continuous	Guinea pig	No significant hematological effect or microscopic lung alterations	Alarie et al., 1973a, 1975
0.38 mg/m³ H_2SO_4 (1.15 μm, MMD), 0.48 mg/m³ H_2SO_4 (0.54 μm, MMD), 2.43 mg/m³ H_2SO_4 (3.6 μm, MMD), or 4.79 mg/m³ H_2SO_4 (0.73 μm, MMD)	78 wk, continuous	Monkey	No significant hematological effect. Morphological changes in the lungs at the two highest concentrations, regardless of the particle size. Alveolar walls were thickened with 2.43 mg/m³, but not 4.79 mg/m³. Hyperplasia and bronchiole thickening only with larger size (1.15 μm, 0.38 mg/m³).	Alarie et al., 1973a
0.89 mg/m³ (90% <0.5 μm in diameter) H_2SO_4	21 h/day, 620 days	Dog	No morphological changes	Lewis et al., 1973
25 mg/m³ H_2SO_4 (1 μm, MMD, $\sigma_g = 1.6$)	6 h/day, 2 days	Guinea pig	Segmented alveolar hemorrhage, type 1 pneumocyte hyperplasia, and proliferation of pulmonary macrophages	Cockrell and Busey, 1978
50 mg/m³ H_2SO_4, or 100 mg/m³ H_2SO_4, or 200 mg/m³ H_2SO_4, or any of of these doses combined with 5 mg/m³ carbon particles (at all three duration schedules)	3 h/day, 20 days; or 3 h/day, 10 days; or 3 h/day, 5 days	Mouse	Damage proportional to concentration, not duration	Ketels et al., 1977

Silicon is ubiquitous in the earth's crust. Silicon dioxide (SiO_2) is found in three crystalline forms (quartz, cristobalite, and tridymite), whose toxicity is ranked tridymite >cristobalite > quartz. While these uncombined forms of SiO_2 are generally called "free silica," SiO_2 combined with cations is called silicate(s). Several hypotheses of the etiology of silicosis have been developed, but no single one has been proven definitively. According to one widely accepted hypothesis developed from both animal and human studies, alveolar macrophages ingest the particles, die, and release their intracellular contents, including lysosomal enzymes and SiO_2. This is followed by a recycling of particle ingestion by macrophages and their death, slow accumulation of other macrophage cells, increased collagen synthesis in response to macrophage lysosomal enzymes, hyalinization, and perhaps complicating factors. Since the alveolar macrophage hypothesis does not explain completely the etiology and pathogenesis of the disease, it is likely that additional factors contribute to the disease. These might include autoimmunity, coexisting tuberculosis or other infections, and/or alterations of lung lipid content and metabolism.

Many animal toxicological studies of SiO_2 exist. Unfortunately, comparisons are difficult because of the species and strain of animal, accidental infections, and the size and crystalline form of SiO_2 particle used.

Silicosis similar to that observed in man has been produced in animals exposed to high concentrations of quartz and other SiO_2 dusts via intratracheal instillation (30-50 mg) or chronic inhalation. Chronic exposures (2.5 yr) of dogs to diatomaceous earth containing 61 percent cristobalite produced fibrotic nodules in hilar lymph nodes, but not the lungs.

Several studies of the hemolysis of red blood cells by particles have been reported. This model may correlate with the ability of mineral dusts to cause lung fibrosis *in vivo* and thus is used for screening. Ottery and Gormley (1978) studied the influence of particle size of quartz (Min-u-sil) and other materials on red cell hemolysis. The particle size of the quartz ranged from 2.7 to 6.8 µm (mean volume diameter, MVD). At lower concentrations (0.025 to about 0.15 mg/ml), there was a linear dose-response increase in hemolysis for quartz (1.35 and 3.55 µm, MVD), kaolin (4.7 µm, MVD), cristobalite (3.05 µm, MVD), and bentonite (5 µm MVD). In this experiment, the effectiveness for increasing hemolysis was ranked as bentonite >kaolin > quartz > cristobalite. Even though this test typically is used to predict fibrotic potential, it should be noted that cristobalite usually is more fibrogenic than quartz, which is much more fibrogenic than the silicates. In another experiment, kaolin and quartz were additive when mixed. Linear increases in hemolysis were also observed with increasing numbers of particles. When the various sizes of Min-u-sil were directly compared, as particle size decreased from 6.8 to 2.7 µm, a smaller concentration was required to produce 5 percent hemolysis. For example, the largest size tested required 2.7 mg/ml, whereas only 0.21 mg/ml was needed for the smallest size tested.

Kysela et al. (1973) administered high concentrations (50 mg) of 9 sizes of quartz dust (0.7 to 35 μm) to rats by intratracheal instillation. A variety of biochemical determinations as well as a histological examination of the lungs were made 3 mo after dosing. As particle size decreased, there was a trend towards increased wet weight of the lung, hydroxyproline content, total lipids, esterified fatty acids, and phospholipids. Cholesterol showed only a slight increase. For hydroxyproline in total lung, the increase was in steps, with increments at about 0.9, 5, 7, and 10 μm. Between 0.7 and 14 μm, the increase was significant ($p < 0.05$). Lipid changes, per gram of tissue, exhibited a trend towards linearity with decreasing particle size. The larger particles (14 to 35 μm) caused a stationary granulamatous response. With the intermediate particles (5 to 10 μm), the lungs had cellular nodules with a few collagenous fibers, increased tissue cellularity, and endoalveolar foam cells. With the smaller particles, the nodules were more numerous and collagenous.

Goldstein and Webster (1966) also investigated the effects of size-graded quartz particles in rats exposed by intratracheal instillation and examined 4 mo later. The sizes and amounts used (<1 μm, 13.99 mg; 1 to 3 μm, 46.1 mg; and 2 to 5 μm, 92.7 mg) were such that the rats were exposed to an equivalent surface area (600 sq. cm.) for each of the size ranges. The < 1 μm particles caused more numerous nodules. The two other size ranges produced an equivalent number of lungs with nodules, but there were many more lungs with confluent nodules, compared to the smallest size quartz. The degree of fibrosis was similar in the 1 to 3 μm and 2 to 5 μm groups and was more severe than that observed in the < 1 μm group. The weight of collagen in lungs increased as particle size increased. It should be recalled, however, that the concentration (weight per exposure) of particles was increased as particle size increased.

In vitro studies (Mossman et al., 1978) with small carbon particles (0.5 to 1 μm) suggest that particles deposited in the TB region that become attached to the mucosal surfaces of the airways may subsequently enter the submucosa and be taken up by mesenchymal cells of the trachea.

12.3.3 Alterations in Pulmonary Function

12.3.3.1 Acute Exposure Effects--On short-term exposure, respiratory mechanics are very sensitive to inhaled H_2SO_4 and some other compounds. Amdur (1971) has cautioned that her method for measuring airway resistance (Amdur and Mead, 1955, 1958) should not be used as an indication of chronic toxicity and should be considered only for short-term toxicity. The Mead-Amdur method uses unanesthetized guinea pigs in which a transpleural catheter has been implanted. Amdur (1971) suggests that, if anything, this procedure increases rather than decreases the sensitivity of the guinea pigs to inhaled irritants.

Using this method, Amdur and coworkers (Amdur and Underhill, 1968, 1970; Amdur, 1954; 1958, 1959, 1961; Amdur and Corn, 1963; Amdur et al., 1978a,b,c,) have studied the effects of aerosols alone (see Table 12-6) or in combination with SO_2 (see Section 12.4.1.1). In all of their studies, exposures were for 1 hour. The method records resistance to air flow in and

TABLE 12-6. RESPIRATORY RESPONSE OF GUINEA PIGS EXPOSED FOR 1 HOUR TO PARTICLES IN THE AMDUR et al. STUDIES

Compound	Concentration mg/m^3	Particle size, μm, MMD	Resistance cm H_2O/ml/sec % difference from control	Compliance ml/cm H_2O % difference from control	Reference
H_2SO_4	0.10	0.3	+41[a]	-27[a]	Amdur et al., 1978b; Amdur, 1977
	0.51	0.3	+60[a]	-33[a]	Amdur et al., 1978b
	1.00	0.3	+78[a]	-40[a]	Amdur et al., 1978b
	1.90	0.8	+51[a]	-35[a]	Amdur, 1969; Amdur, 1958
	5.30	0.8	+54[a]	-40[a]	Amdur, 1958
	15.40	0.8	+69[a]	-24[a]	Amdur, 1958
	26.1	0.8	+89[a]	-38[a]	Amdur, 1958
	42.00	0.8	+120[a]	-26[a]	Amdur, 1958
	0.11	1.0	+14[a]	-13	Amdur et al., 1978b
	0.40	1.0	+30[a]	-8	Amdur et al., 1978b
	0.69	1.0	+47[a]	-25[a]	Amdur et al., 1978b
	0.85	1.0	+60[a]	-28[a]	Amdur et al., 1978b
	2.30	2.5	+39[a]	-16	Amdur, 1958
	8.90	2.5	+61[a]	-26[a]	Amdur, 1958
	15.40	2.5	+96[a]	-43[a]	Amdur, 1958
	43.60	2.5	+317[a]	-76[a]	Amdur, 1958
	30.50	7.0	+42[a]	-17	Amdur, 1958
$(NH_4)_2SO_4$	0.50	0.13	+23[a]	-27[a]	Amdur et al., 1978a
	2.14	0.20	-4	-13[a]	Amdur et al., 1978a
	1.02	0.30	+29[a]	-23[a]	Amdur and Corn, 1963; Amdur et al., 1978a; Amdur, 1974
	9.54	0.81	0	-12[a]	Amdur et al., 1978a
NH_4HSO_4	0.93	0.13	+15[a]	-15[a]	Amdur et al., 1978a
	2.60	0.52	+28[a]	-30[a]	Amdur et al., 1978a
	10.98	0.77	+23[a]	-19[a]	Amdur et al., 1978a

TABLE 12-6. (continued).

Compound	Concentration mg/m^3	Particle size, μm, MMD[b]	Resistance cm H_2O/ml/sec % difference from control	Compliance ml/cm H_2O % difference from control	Reference
Na_2SO_4	0.90	0.11	+2	-7	Amdur et al., 1978a
$ZnSO_4$	0.91	1.4	+41[a]	-	Amdur and Corn, 1963; Amdur, 1974
$ZnSO_4 \cdot (NH_4)_2SO_4$	0.25	0.29	+22[a]	-	Amdur and Corn, 1963; Amdur, 1977
	0.50	0.29	+40[a]	-	Amdur and Corn, 1963
	1.10	0.29	+81[a]	-	Amdur and Corn, 1963; Amdur, 1974
	1.80	0.29	+129[a]	-	Amdur, 1969; Amdur and Corn, 1963
	1.50	0.51	+43[a]	-	Amdur and Corn, 1963; Amdur, 1977
	2.48	0.51	+68[a]	-	Amdur and Corn, 1963
	1.40	0.74	+29[a]	-	Amdur, 1969; Amdur and Corn, 1963; Amdur, 1977
	1.10	1.4	+6		Amdur and Corn, 1963; Amdur, 1977
	3.60	1.4	+32[a]		Amdur and Corn, 1963
$CuSO_4$	0.43	0.11	+9	-11[a]	Amdur et al., 1978a
	2.05	0.13	+25[a]	-15[a]	Amdur et al., 1978a
	2.41	0.33	+14[a]	-11[a]	Amdur et al., 1978a
$NaVO_4$	0.70	-	+7	-	Amdur and Underhill, 1968
$FeSO_4$	1.00	-	+2	-	Amdur and Underhill, 1968

TABLE 12-6. (continued)

Compound	Concentration mg/m^3	Particle size, µm, MMD	Resistance cm H_2O/ml/sec % difference from control	Compliance ml/cm H_2O % difference from control	Reference
Fe_2O_3 (2hr)	11.70	0.076 (GMD)	-9	5	Amdur and Underhill, 1968; 1970
(Fumes)	21.00	0.076 (GMD)	0	0	Amdur and Underhill, 1968; 1970
$MnCl_2$	1.00	-	+4	-	Amdur and Underhill, 1968
MnO_2	9.70	-	-6	-	Amdur and Underhill, 1968
$MnSO_4$	4.00	-	-1	-	Amdur, 1974
Open hearth	0.16	0.037 (GMD)	+9	0	Amdur and Underhill, 1968; 1970
dust	7.00	0.037 (GMD)	+6	-16	Amdur and Underhill, 1968; 1970
Activated carbon	8.70	-	-3	-	Amdur and Underhill, 1968
Spectographic	2.00	-	+7	-	Amdur and Underhill, 1968
carbon	8.00	-	+17	-	Amdur and Underhill, 1968

[a] $p < 0.05$

[b] Diameters are provided as mass median diameter (MMD) unless specified as geometric median diameter by count (GMD).

out of the lungs and airways, compliance (a measure of lung distensibility), tidal volume (the volume of air moved during normal breathing), respiratory frequency, and minute volume. While increased flow resistance is often the most striking feature of the response to aerosols, calculations of the elastic, resistive, and total work of breathing can also be made. The method is, therefore, nearly as elaborate and inclusive an evaluation of pulmonary mechanics as could be made in small laboratory animals until Drazen's (1976) work.

The importance of particle size on the site of pulmonary deposition is described in Chapter 11. Sulfuric acid aerosols ranging in concentration from 1.9 to 43.6 mg/m^3 were generated in three particle sizes: 0.8 µm (σ_g = 1.32), 2.5 µm (σ_g = 1.38), or 7 µm (σ_g = 2.03) MMD. Particles of the largest size (7 µm, at 30 mg/m^3) produced a significant increase in flow resistance, but no other detectable changes in respiration. At the lowest concentration tested (1.9 mg/m^3), the 0.8 µm particles increased the resistance to flow and the elastic, resistive, and total work of breathing; but they produced a decrease in compliance. The 2.5 µm particles also increased the resistance to flow at concentrations from 2.3 to 43.6 mg/m^3. The relative efficacy of the 0.8 and 2.5 µm particles differed. At concentrations of 2 mg/m^3, the 0.8 µm particles were more effective than the 2.5 µm particles. The time course of the response also varied with the particle size, since the 2.5 µm particles did not produce their major effects until the last 15 to 20 min of the 1-h exposure. These differences in response were probably associated with the degree and site of constriction within the bronchi. The 2.5 µm particles produced constriction in the larger bronchi, whereas the 0.8 µm particles caused narrowing of the smaller bronchi. In earlier work, Pattle et al. (1956) reported that, at higher concentrations, 2.7 µm aerosols are more toxic than 0.8 µm as measured by mortality and increased airway resistance. While the results of the experiments by Amdur and coworkers are reported in a straightforward concentration-response curve, the physiological mechanisms producing the measurable effects are obviously highly complex. Detailed understanding is lacking.

Amdur et al. (1978b) exposed guinea pigs for 1 h to either 0.3 or 1 µm (MMD) H_2SO_4 in concentrations ranging from 0.1 to 1 mg/m^3. The concentration-response ratio for percent change in resistance was linear for both particle sizes; however, the smaller particle caused a greater response, particularly at 0.1 mg/m^3, where a 26 percent increase in airway resistance was observed. All increases in resistance were statistically significant. The smaller particle size also decreased compliance at all concentrations tested; however, the lowest effective concentration tested for the 1 µm particle was 0.69 mg/m^3. For equivalent concentrations, the 0.3 µm particle decreased compliance more than the 1 µm particle. Animals were also examined 30 min after exposure. After exposure to 0.1 mg/m^3 H_2SO_4 (0.3 µm), resistance was still elevated above control in guinea pigs; but for the 1 µm particle, recovery had occurred. These exposures caused no alterations of tidal volume, respiratory frequency, or minute volume. In comparing these results to earlier work with SO_2 (Amdur, 1966), Amdur et al. (1978b) describe how the same amount of sulfur, when given as H_2SO_4, produces 6 to 8 times the response observed than when given as SO_2.

Silbaugh et al. (1981) exposed Hartley guinea pigs for 1 h to 1 µm (MMAD) H_2SO_4 aerosols at concentrations and relative humidities (RH) of 0 mg/m^3 (control group) (40 or 80 percent RH), 1.2 mg/m^3 (40 percent RH), 1.3 mg/m^3 (80 percent RH), 14.6 mg/m^3 (80 percent RH), 24.3 mg/m^3 (80 percent RH), or 48.3 mg/m^3 (80 percent RH). Ten animals were exposed at each concentration except for the 24.3 and 48.3 mg/m^3 groups, which consisted of 9 and 8 animals, respectively. Measurements of tidal volume, breathing frequency, minute volume, peak inspiratory and expiratory flow, tidal transpulmonary pressure excursions, total pulmonary resistance, and dynamic lung compliance were obtained every 15 min during (1) a 30-min baseline period, (2) the 60-min exposure period, and (3) a 30-min recovery period. Pulmonary function changes in H_2SO_4-exposed guinea pigs did not differ from controls, except for 1 animal exposed to 14.6 mg/m^3, 3 animals exposed to 24.3 mg/m^3, and 4 animals exposed to 48.3 mg/m^3. Pulmonary function changes in these 8 responsive animals included marked increases in total pulmonary resistance and marked decreases in dynamic compliance. Two animals in each of the latter groups died during exposure. The proportion of responsive to nonresponsive animals increased with exposure concentration, but the magnitude of pulmonary function change was similar for all responsive animals. Compared to nonresponders, responsive animals tended to have higher preexposure values of total pulmonary resistance and lower preexposure values of dynamic compliance. The authors suggested that guinea pigs react to acute H_2SO_4 exposure with an essentially all-or-none airway constrictive response. The finding that resistance and compliance changes are important components of the guinea pig's airway response to H_2SO_4 aerosols is consistent with Amdur et al. (1978b). The presence of high preexposure pulmonary resistance values in responsive animals is similar to the finding (Amdur, 1964) that guinea pigs with high preexposure resistance values were those most severely affected during irritant aerosol exposure. The lack of effects at lower concentrations and the essentially all-or-none airway constrictive response observed in these studies, however, differs markedly from the graded response observed by Amdur et al. (1958, 1978b) during similar exposures. The reasons for the differences in experimental results are unclear, but may be at least partially related to differences in animal strains and techniques. These results indicate that changes in respiratory function do not occur at environmental concentrations of H_2SO_4 in most animals, but suggest that susceptible subpopulations might possibly exist in some species. (See discussion above about susceptible individuals.)

It should be noted, however, that Sackner et al. (1978a,b) evaluated pulmonary function in anesthetized dogs either immediately or 2 h after exposure to approximately 18 mg/m^3 H_2SO_4 for 7.5 min or to 4 mg/m^3 H_2SO_4 for 4 hours. The MMAD was < 0.2 µm. There were no significant changes in respiratory resistance, specific respiratory conductance, specific lung compliance, or functional residual capacity. At the higher concentration, cardiovascular parameters (e.g., blood pressure, cardiac output, heart rate, and stroke volume) and arterial blood gas tensions were also studied, but no significant changes were observed. In addition, the pulmonary function (pulmonary resistance and dynamic compliance) of donkeys was not affected by H_2SO_4 exposure (1.51 mg/m^3, 0.3 to 0.6 MMAD, 1 h) (Schlesinger et al., 1978, 1979). The small size of the particles may be responsible for the lack of an effect. Larger particles may be more potent, since resistance and compliance measurements reflect alterations of the larger conducting airways in the TB region of deposition.

Studies of the irritant potential of sulfate salts have shown that these aerosols are not innocuous, but evoke increased flow resistance similar to H_2SO_4 aerosols. The influence of particle size on the effects of zinc ammonium sulfate has also been investigated by Amdur and Corn (1963). They showed in guinea pigs exposed for 1 h that zinc sulfate had about half the potency of zinc ammonium sulfate, with ammonium sulfate being one-third to one-fourth as potent as zinc ammonium sulfate. Zinc ammonium sulfate was chosen for study because it was reported as a major component of the aerosol from the severe air pollution episode in Donora, Pa. in 1948 (Hemeon, 1955). Zinc ammonium sulfate is not a species commonly found in urban air. Four sizes of aerosols were administered: 0.29, 0.51, 0.74, and 1.4 µm (particle mean size by weight). When the aerosol concentration was held constant at 1 mg/m^3, the smaller particles produced greater increased resistance to flow. This response was thought to be the result of the number of particles rather than of differential sites of deposition. The dose-response curve also became steeper with decreasing particle size. These data should be compared carefully with those from similar human exposures (Chapter 13, Sect. 13.5.2).

Amdur et al. (1978a) compared the effects of $(NH_4)_2SO_4$, NH_4HSO_4, $CuSO_4$, and Na_2SO_4. Although particle sizes and concentrations were not matched precisely throughout the study, statistical analyses for ranking were not applied, and the degree of response increased with decreased size (size range, 0.1 to 0.8 µm, MMD), the authors suggest that the order of irritant potency was $(NH_4)_2SO_4 > NH_4HSO_4 > CuSO_4$. Sodium sulfate (0.11 mg/m^3, 0.11 MMD) caused no significant effects on either resistance or compliance. At the lowest concentrations used, $(NH_4)_2SO_4$ (0.5 mg/m^3, 0.13 µm MMD), NH_4HSO_4 (0.93 mg/m^3, 0.13 µm MMD), and $CuSO_4$ (0.43 mg/m^3, 0.11 µm MMD) decreased compliance. These concentrations of $(NH_4)_2SO_4$ and NH_4HSO_4 also increased resistance. For $CuSO_4$, the lowest concentration tested that caused an increase in resistance was 2.05 mg/m^3 (0.13 µm MMD). All of these compounds are less potent than H_2SO_4 in the Amdur studies.

Comparisons between H_2SO_4 and sulfate salt aerosols are difficult because of the marked dependence of the efficacy on the aerosol size. If the particles are of identical size, H_2SO_4 is more efficacious than zinc ammonium sulfate; but if the zinc ammonium sulfate were present as a submicron aerosol and the H_2SO_4 as a large aerosol, then zinc ammonium sulfate would be more efficacious at the same concentration (Amdur, 1971). Regardless of the particle size, the equivalent amount of sulfur present as SO_2 is much less efficacious than if it were present as a sulfate salt containing ammonium or H_2SO_4. When present as SO_2, 2.62 mg/m^3 (1 ppm) SO_2 is equivalent to 1.3 mg/m^3 sulfur and produces a 15 percent increase in flow resistance. If this amount of sulfur were present as a 0.7 µm aerosol of H_2SO_4, it would result in a 60 percent increase in flow resistance or be about 4 times more efficacious. If the sulfur were present as zinc ammonium sulfate in a 0.3 µm aerosol, the increase in flow resistance would be about 300 percent (20-fold) in efficacy. Some sulfate salt aerosols are not irritating. For example, though ferrous sulfate and manganous sulfate do not cause an increase in flow resistance, ferric sulfate does cause this response. A summary of irritant potency is presented below.

Relative Irritant Potency of Sulfates In Guinea Pigs
Exposed for 1 Hour[a] (Amdur et al., 1978a)

Sulfuric acid	100
Zinc ammonium sulfate	33
Ferric sulfate	26
Zinc sulfate	19
Ammonium sulfate	10
Ammonium bisulfate	3
Cupric sulfate	2
Ferrous sulfate	0.7
Sodium sulfate (at 0.1 µm)	0.7
Manganous sulfate	-0.9

[a] Data are for 0.3 µm (MMD) particles. Increases in airway resistance were related to sulfuric acid (0.41 percent increase in resistance per µg of sulfate per m^3 as sulfuric acid) that was assigned a value of 100.

Nadel et al. (1967) found that zinc ammonium sulfate (no concentration given) and histamine aerosols produced similar increases in resistance to flow and decreases in pulmonary compliance in cats. Histamine was more potent than zinc ammonium sulfate. The increase in flow resistance could not be blocked by intravenous administration of atropine sulfate, but was blocked by either intravenous or inhaled isoproterenol. The study suggested that the increased flow resistance was due to an increase in bronchial smooth muscle tone. Histamine appears to be a likely mediator of the bronchoconstriction following inhalation of sulfate salt aerosols. Charles and Menzel (1975a) investigated the release of histamine from guinea pig lung fragments incubated with varying concentrations of sulfate salts. Almost complete release of tissue histamine occurred with 100 mM ammonium sulfate. Intratracheal injection of ammonium sulfate also released all of the histamine from perfused and ventilated rat lungs (Charles et al., 1977a). The potency ranking of different sulfate salts in the release of histamine from lung fragments (Charles and Menzel, 1975a; Charles et al., 1977a) was equivalent to that causing increased resistance to flow (Amdur et al., 1978a). Bronchoconstriction of the perfused lung occurred on intratracheal injection of sulfate salts or histamine (Charles et al., 1977a). About 80 percent of the constriction could be blocked by prior treatment of the isolated lungs with an H-1 antihistamine. These experiments, as well as the original observations of Nadel et al. (1967) and Amdur et al. (1978a), support the concept that an intermediary release of histamine or some other vasoactive hormone is involved in the irritant response of sulfate aerosols. An ammonium sulfate particle is calculated to reach a concentration of about 275 mM at equilibration with the 99.5 percent RH of the respiratory tract (National Research Council, 1978). Thus, the concentration of the hydrated particle on striking the mucosa would be within the range found to cause release of histamine in guinea pig and rat lung fragments (Charles and Menzel, 1975a; Charles et al., 1977a). A published estimate of the dose of inhaled ammonium sulfate needed to release histamine in the lung is in error (National Research Council, 1978). Complete release of histamine (100 percent) occurred with 1 μmole of ammonium sulfate/lung, and not a 1 μM solution for the entire lung (Charles et al., 1977a). Further, total release of all histamine stores of a tissue rarely, if ever, occurs under physiological conditions. Only about 10 percent of the total histamine is released during degranulation reactions _in vivo_, producing anaphylactic shock and death. Therefore, even if the calculations were correct, only a small fraction of the ammonium sulfate dose would be required to produce the far less violent increases in flow resistance reported by Amdur et al. (1978a) for ammonium bisulfate and ammonium sulfate. Assuming the calculation of ammonium sulfate to be correct, a 4-h, not a 2-day, inhalation would produce a marked increase in resistance to flow. Additionally, Charles et al. (1977b) found the rate of removal of ${}^{35}SO_4^{-2}$ from the rat lung both _in vivo_ and _in vitro_ to be a function of the cation associated with the salt and to follow the same order of potency as reported by Amdur and coworkers (1978a) in the guinea pig irritancy test. Especially noteworthy is the fact that manganous sulfate was removed at essentially the same rate as sodium sulfate, both of which did not produce increased resistance to flow in the guinea pig.

Hackney (1978) has presented a preliminary summary of the effects of aerosols of H_2SO_4 and nitrate and sulfate salts on squirrel monkeys (_Saimiri sciurens_). Monkeys were exposed (head only) to aerosols (2.5 mg/m^3) of the respective salts or H_2SO_4 (40 or 85 percent RH at 25°C). The exposure system was designed to reduce stress on the unanesthetized monkey. A noninvasive method of pulmonary function measurement was used in which total respiratory resistance was measured by the forced pressure oscillation technique at sine wave frequencies of either 10 or 20 Hz. The measurement of pulmonary resistance included the resistance of the chest wall, which was assumed to be irrelevant to pollutant response and to be constant throughout the experiments. To correct for stress, control values were taken as those for a given monkey exposed on the previous day to an aerosol of distilled water (for aerosol experiments).

Hackney (1978) reports that the measurement of respiratory resistance was frequency-dependent, with changes in resistance appearing greater in the 10 than the 20 Hz measurement frequency. (The measurement frequency is not to be confused with the breathing frequency.) The exposure period in the experiments was either 1 or 2 hours. Some aerosols were studied only at 40 percent RH. No attempt was made to define a dose-response curve for aerosols, and all exposures were at or near 2.5 mg/m^3. At low RH (40 percent RH; MMAD 0.3 µm, $\sigma_g = 2.0$), there were no differences between $(NH_4)_2SO_4$-exposed and control values, while at high relative humidity (85 percent RH, MMAD 0.6 µm, $\sigma_g = 2.3$), 3 of 5 monkeys had increased airway resistance by 1 hour. Zinc ammonium sulfate aerosols produced increased resistance at low humidity (40 percent RH; MMAD 0.3 µm, $\sigma_g = 2.5$) but no consistent increases over control values at high humidity (85 percent RH; MMAD 0.6 µm, $\sigma_g = 1.6$). Ammonium bisulfate (40 percent RH; MMAD 0 4 µm, $\sigma_g = 1.8$) also produced increased resistance at 2.7 mg/m^3.

Hackney's (1978) data from exposures to H_2SO_4 and ammonium nitrate aerosols were analyzed by computer and differed quantitatively from the data reported above for those exposures that were reduced by hand. Differences were probably due to a systematic error in the hand-reduced data that required a judgement in selection of raw data points. The biological interpretation does not appear to be altered by these two approaches, but it does point out the experimental difficulties in interpretation of pulmonary function data from experimental animals. While H_2SO_4 aerosols (40 percent RH; MMAD 0.4 µm, $\sigma_g = 2.0$) caused no statistically significant increases, there was a trend toward increased resistance after 60 min that then tended to decline. Ammonium nitrate exposures produced no changes.

Multiple contrast statistical analysis of the Hackney (1978) data revealed no significant differences between baseline or control values for any exposure using data collected at the 20 Hz measurement frequency. At the 10 Hz measurement frequency, the data were more variable, but significant differences indicative of increased airway resistance could be found for animals exposed to $ZnSO_4$, $(NH_4)_2SO_4$ and H_2SO_4 at 40 percent RH. Several procedural aspects should be recognized. First, data were analyzed on a group mean basis, even though large differences between individual monkeys existed in both variability and absolute magnitude. Second, the time course of exposure to the aerosols illustrated a trend indicative of a transient response by the monkeys to sulfate, nitrate, or H_2SO_4 aerosols. The use of group means ended to reduce the magnitude of the response and flatten the response-time curve. This is certainly true for the SO_2 exposures. Third, there were major differences in the response measured at either 10 or 20 Hz. Fourth, the response estimated by both manual and computer reduction differed by as much as 40 percent. Compared to the data reported for guinea pigs, however, these experiments support the general trends originally proposed from the guinea pig data.

Larson and coworkers (1977, 1982) have proposed that breath ammonia is important in neutralizing inhaled H_2SO_4. Ammonia is released in the breath from blood ammonia and bacterial decay products in the buccal cavity. Ammonia in the breath could react with H_2SO_4 to produce ammonium bisulfate or ammonium sulfate, depending on the amount of ammonia and H_2SO_4 present in the aerosol droplet. Complete neutralization of H_2SO_4 produces ammonium sulfate ($(NH_4)_2SO_4$). This theory has been discussed at some length (National Research Council, 1978). Much of the data has not yet been published, so a critical review of the model given for the neutralization of H_2SO_4 aerosol droplets by gaseous ammonia is not available. However, it does appear that neutralization is inversely proportional to particle size (Larson et al., 1982).

The biological effects of H_2SO_4 aerosols could be due to a combination of factors. First, the pH of the particle could be very important. Larson et al. (1977, 1982) have calculated the neutralization capacity of the breath ammonia. Once the neutralization capacity of the ammonia present in the breath is exceeded, the pH of the aerosol reaching the lung may fall rapidly. At low pH, physical properties of the mucous layer lining the upper airways may be altered (Holma et al., 1977) or the permeability of the lung may be increased (Charles, 1976). Second, the chemical composition of the sulfate aerosol, if other than H_2SO_4, may also alter the permeability of the lung to sulfate (Charles et al., 1977b; Charles, 1976; Charles and Menzel, 1975b). Third, the cation associated with the sulfate compound may have pharmacological properties in and of itself. The permeability of the lung to sulfate ion presented as various sulfate salts (Charles et al., 1977b) is in the same relative order as the irritant potential found for aerosols of the same sulfate salts (Amdur et al., 1978a).

It is likely that ammonia functions within pulmonary tissue as a source of protons that increases the flux of sulfate to the site of action. Ammonia can diffuse readily across cell membranes to react with protons, forming ammonium ion. Intracellular transport of negatively charged sulfate would result in the concomitant accumulation of protons to preserve electrochemical neutrality. At physiological pH values, a significant fraction of ammonium salts is present as ammonia. Ammonium salts could augment the local ammonia concentration and thus increase the uptake of sulfate ions and result in release of histamine. Ammonia increased uptake of sulfate by the lung (Charles et al., 1977a; Charles, 1976), possibly by this mechanism.

In relation to H_2SO_4, ammonium sulfate and bisulfate are less irritating to the lung because of their higher pH values once dissolved in the milieu of the lung. Thus, neutralization of H_2SO_4 aerosols by breath ammonia could be an important detoxification step. The concept of neutralization by breath ammonia does not negate the histamine release hypothesis, since ammonium sulfate is active in the release of histamine in guinea pig lung fragments (Charles, 1976) and in rat lungs (Charles and Menzel, 1975b).

An important problem is the relationship of these observations to human health effects. Unfortunately, histamine release by nonimmune-mediated reactions, such as the apparent ion exchange process due to sulfate interaction with mast cell granules (Charles, 1976), is poorly understood. Metabolism of histamine by man and rodents could have important different implications. Also, not all of the pharmacological action of ammonium sulfate instilled intratracheally in the perfused rat lung could be blocked by an H-1 antihistamine (Charles et al., 1977a). A number of other inflammatory hormones, aside from histamine, mediate bronchial tone in man. Slow reacting substances of anaphylaxis (SRS-A or leukotrienes), prostaglandins, and kinins would not be blocked by an H-1 antihistamine. Thus, species differences are not unanticipated, but should be clarified so that the potential applicability of these data to man is understood.

The biological effect of sulfate compounds is highly dependent on the chemical composition of the compound. For example, H_2SO_4 has much more potent effects on pulmonary function than any sulfate salt, but the sulfate salts also have differing potency. The cations associated with the sulfate ion may promote its transport, thereby increasing the biological response. The cation has biological effects by itself; as discussed here. It is not possible, then, to predict the potential toxicity of a sulfate aerosol based solely on the sulfate content. Clearly, the acidity of the aerosol plays an important role in the toxicity, as do particle size and other physical properties.

An important experimental problem is raised by the ammonia neutralization of H_2SO_4. Ammonia is produced in all animal experimental exposure systems through the accumulation of urine and feces (Malanchuk et al., 1980). This is particularly so in whole-body chronic exposures. Few exposure systems provide a rapid turnover of the chamber air (e.g., 1 chamber volume/min), and given the technological problems in monitoring ammonia, even this rate of airflow may be insufficient. The usual turnover rate is 10 to 15 chamber volumes of air/h or less. Under these conditions, animals exposed to H_2SO_4 aerosols may, in fact, be inhaling ammonium sulfate and ammonium bisulfate aerosols as well. The high concentrations of H_2SO_4 aerosols needed to produce significant pathological effects during chronic exposure may be due to these chemical conversions. The level of ammonia in the breath of animals is also unknown and is sure to vary with the diet of the animals. Some commercial animal diets are low in protein, while others are high. The blood ammonia will depend, in part, on the total amount of protein and quality of the protein, as well as on the kidney function of the animal. What effects, if any, the buccal flora have on the exhalation of ammonia in animals is totally unknown. Certainly, the propensity of SO_2- and H_2SO_4-exposed animals to develop nasal infections raises disturbing questions. The ability of the buccal flora of animals to produce ammonia may be very different from humans. This technical problem of ammonia in the exposure atmosphere should be addressed and solved in both human and animal exposures before further reliance can be placed on these data for H_2SO_4 (Table 12-7).

12.3.3.2 Chronic Exposure Effects--The influence of chronic exposure to H_2SO_4 on pulmonary function was investigated by Alarie et al. (1973a, 1975). Guinea pigs exposed continuously to either 0.9 mg/m^3 (0.49 µm, MMD) (Alarie et al., 1975), 0.1 mg/m^3 (2.78 µm, MMD) (Alarie et al., 1973a), or 0.08 mg/m^3 (0.84 µm, MMD) for 52 wk (Alarie et al., 1973a) had no significant changes of pulmonary mechanics (including measurements of flow resistance, respiratory rate, lung volumes, and work of breathing) that could be attributed to H_2SO_4. Cynomolgus monkeys exposed continuously and tested periodically during 78 wk, however, were affected by some treatment regimens (Alarie et al., 1973a). Monkeys exposed to 0.48 mg/m^3 (0.54, µm, MMD) experienced an altered distribution of ventilation (increased nitrogen washout) early in the exposure period, but recovery occurred during exposure. Animals exposed to a similar concentration (0.38 mg/m^3) but a larger particle size (2.15 µm, MMD) had no change in this parameter. Higher concentrations altered distribution of ventilation, with the lesser concentration

TABLE 12-7. EFFECTS OF ACUTE EXPOSURE TO SULFATE AEROSOLS ON PULMONARY FUNCTION[a]

Concentration	Duration	Species	Results	Reference
0 mg/m^3 (40 or 80% RH), 1.2 mg/m^3 (40% RH), 1.3 mg/m^3 (80% RH), 14.6 mg/m^3 (80% RH), 24.3 mg/m^3 (80% RH), and 48.3 mg/m^3 (80% RH) 1 μm (MMAD) H_2SO_4	1 h	Guinea pig	Pulmonary function changes observed in one animal (out of 10) exposed to 14.6 mg/m^3, three animals (out of 9) exposed to 24.3 mg/m^3, and four animals (out of 8) exposed to 48.3 mg/m^3.	Silbaugh et al., 1981
0.8 - 1.51 mg/m^3 H_2SO_4 (0.3 - 0.6 μm, MMAD) or 0.4 - 2.1 mg/m^3 $(NH_4)_2SO_4$ (0.3 - 0.6 μm, MMAD)	1 h	Donkey	No significant alterations in pulmonary resistance and dynamic compliance	Schlesinger et al., 1978
2.5 mg/m^3 $(NH_4)_2SO_4$, $ZnSO_4$,$(NH_4)_2SO_4$, H_2SO_4, and NH_4NO_3, 2.7 mg/m^3 NH_4HSO_4	1 h	Monkey	Increased airway resistance at high relative humidity for $(NH_4)_2SO_4$, and low relative humidity for $ZnSO_4$, $(NH_4)_2SO_4$. NH_4HSO_4 also increased resistance. No significant effects with H_2SO_4 or NH_4NO_3	Hackney, 1978

[a] See Table 12-6 for the Amdur et al. studies on pulmonary function effects in guinea pigs.

($2.43 mg/m^3$) and larger particle size (3.6 µm, MMD) causing an onset sooner (at 17 compared to 49 wk) than in monkeys exposed to $4.79 mg/m^3$ H_2SO_4 (0.73 µm, MMD). Beginning approximately at 8 to 12 wk of exposure, $0.38 mg/m^3$ (2.15 µm, MMD), $2.43 mg/m^3$ (3.6 µm, MMD) and $4.79 mg/m^3$ (0.73 µm, MMD) H_2SO_4 increased respiratory rate. The only alteration in arterial partial pressure of O_2 was a decrease observed in monkeys exposed to $2.43 mg/m^3$. Except for respiratory rate as described above, mechanical properties (including resistance, compliance, tidal volume, minute volume, and work of breathing) were not altered significantly by the chronic H_2SO_4 exposures. Morphological studies of these animals are described in Section 12.3.2.

Lewis et al. (1969, 1973) performed chronic studies of dogs. The animals were exposed for 21 h/day for 225 or 620 days to $0.89 mg/m^3$ H_2SO_4 (90 percent <0.5 µm in diameter) alone and in combination with SO_2 (see Section 12.4.1.2 for expanded discussion). After 225 days (Lewis et al., 1969), the dogs receiving H_2SO_4 had a significantly lower diffusing capacity for CO than animals that did not receive H_2SO_4. After 620 days of exposure, CO diffusing capacity was still decreased ($p < 0.05$) (Lewis et al., 1973). In addition, residual volume and net lung volume (inflated) were decreased ($p < 0.05$), and total expiratory resistance was increased ($p < 0.05$). Total lung capacity, inspiratory capacity, and functional residual capacity were also decreased ($p = 0.1$). Other pulmonary function measurements were not significantly affected (see Table 12-8).

12.3.4 Alteration in Host Defenses

To protect itself against inhaled microorganisms and inanimate particles, the host has several defense mechanisms. Microbes reaching the gaseous exchange regions of the lung can be phagocytized and killed by alveolar macrophages. Later these macrophages can move to the ciliated airways where they are cleared from the lung, along with other particles that are deposited on the airways, by the mucociliary escalator. Inanimate particles can also be engulfed and removed from the lung by this means (see Chapter 11). Mucociliary clearance is an important defense against both microorganisms and inanimate particles. It is likely then that an impairment of mucociliary clearance might be expressed as increased infections.

12.3.4.1 Mucociliary Clearance--Fairchild et al. (1975b) investigated the influence of a 1-hour exposure to H_2SO_4 on deposition of inhaled nonviable bacteria (*Streptococcus pyogenes*, 2.6 µm MMAD) in guinea pigs. All exposure regimens used caused no significant alterations of breathing frequency, tidal volume, or minute ventilation. After exposure to $3.02 mg/m^3$(1.8 µm CMD), a 60-percent increase ($p < 0.01$) in total pulmonary bacterial deposition and a proximal shift in the total deposition pattern to the nasopharynx were observed. No alteration in deposition was observed in the trachea or lung. After exposure to $0.32 mg/m^3$ (0.6 µm CMD), no significant effect on total or regional deposition was seen. However, at a lower concentration and particle size ($0.03 mg/m^3$, 0.25 µm CMD), the deposition pattern did shift ($p < .05$) to the trachea, but without a significant change in total pulmonary deposition.

TABLE 12-8. EFFECTS OF CHRONIC EXPOSURE TO H_2SO_4 AEROSOLS ON PULMONARY FUNCTION

Concentration	Duration	Species	Results	Reference
0.08 mg/m^3 H_2SO_4 (0.84 µm, MMD) or 0.1 mg/m^3 H_2SO_4 (2.78 µm, MMD)	52 wk, continuous	Guinea pig	No effects on pulmonary function	Alarie et al., 1975, 1973
0.38 mg/m^3 H_2SO_4 (1.15 µm, MMD) 0.48 mg/m^3 H_2SO_4 (0.54 µm, MMD) 2.43 mg/m^3 H_2SO_4 (3.6 µm, MMD) 4.79 mg/m^3 H_2SO_4 (0.73 µm, MMD)	78 wk, continuous	Monkey	Exposure to 2.43 or 4.79 mg/m^3 altered distribution of ventilation (0.48 mg/m^3 transiently altered distribution). Exposure to 0.38, 2.43 or 4.79 mg/m^3 increased respiratory rate.	Alarie et al., 1973
0.89 mg/m^3 H_2SO_4 (90% <0.5 µm in diameter)	21 h/day, 225 or 620 days	Dog	Lower CO diffusing capacity after 225 days. After 620 days, residual volume and net lung volume were also decreased, as were total lung capacity, inspiratory capacity, and functional residual capacity. Total expiratory resistance was increased.	Lewis et al., 1969, 1973

After studying effects of H_2SO_4 on deposition of bacteria, these investigators (Fairchild et al., 1975a) turned their attention to effects on clearance of bacteria. They showed that 4-h exposures to 15 mg/m^3 H_2SO_4 (3.2 µm, CMD) after exposure to a nonviable radiolabeled streptococcal aerosol reduced the rate of ciliary clearance of the bacteria from the lungs and noses of mice. When mice received a 90-min exposure to 15 mg/m^3 H_2SO_4 (3.2 µm, CMD) 4 days prior to the bacterial aerosol, clearance of nonviable bacteria was reduced in the nose, but not in the lungs. Neither regimen affected clearance of viable streptococci. No significant effects were seen at concentrations of 1.5 mg/m^3 H_2SO_4 (0.6 µm, CMD).

Schlesinger et al. (1978) demonstrated that 1-h exposures to 0.3 to 0.6 µm H_2SO_4 mist at concentrations in the range of 0.19 to 1.36 mg/m^3 produced transient slowing of bronchial mucociliary particle clearance in 3 of 4 donkeys tested (only one of these demonstrated this response at 0.19 mg/m^3). In addition, 2 of the 4 donkeys developed persistently slowed clearance after about 6 exposures. These exposures had no effects on regional particle deposition or respiratory mechanics, and corresponding exposures to $(NH_4)_2SO_4$ up to 2 mg/m^3 had no measurable effects. In subsequent experiments (Schlesinger et al., 1979), the 2 animals showing only transient responses and 2 previously unexposed animals were given daily 1-h exposures, 5 days/wk, to H_2SO_4 at 0.1 mg/m^3. Within the first few wk of exposure, all 4 donkeys developed erratic clearance rates (i.e., rates that, on specific test days, were either substantially slower than or faster than those in their preexposure period). The degree and the direction of change in rate, however, differed to some extent in the different animals. These changes may herald subsequent alterations and might possibly represent important low level signals at the detection limit of the method. The 2 previously unexposed animals developed persistently slowed bronchial clearance sometime during the second 3 mo of exposure and during 4 mo of follow-up clearance measurements, while the 2 previously exposed animals had clearance times that consistently fell within the normal range after the first few weeks of exposure. The alterations of clearance observed in 2 initially healthy and previously unexposed animals may be a significant observation, since alteration of normal mucociliary clearance over a period might have important implications. Lippmann et al. (1980, 1981) and Leikauf et al. (1981) have conducted similar experiments in human subjects (see Chapter 13).

Tracheal mucociliary transport rates have been measured in several other animal studies. Sackner et al. (1978a) failed to find significant changes in tracheal mucus velocity following short-term exposures to 14 mg/m^3 (0.12 µm) H_2SO_4 in sheep. Similarly, Schlesinger et al. (1978) saw no effect on tracheal transport in donkeys after 1-h exposures to concentrations up to 1.4 mg/m^3 (0.3 to 0.6 µm MMAD) H_2SO_4. On the other hand, Wolff et al. (1981a) reported a depression in tracheal transport rate in anesthesized dogs exposed for 1 h to 1.0 mg/m^3 (0.9 µm, MMAD, $\sigma_g = 1.4$) that persisted at 1 wk postexposure. Recovery had occurred when the animals were examined again at 5 wk postexposure. Following a 1-h exposure to 0.5 mg/m^3

H_2SO_4, there were slight increases ($p > 0.05$) in tracheal mucous velocities both immediately and 1 day after exposure. One wk after exposure, however, clearance was significantly decreased. The latter results are similar to those observed in the bronchi of individual humans in the Lippmann et al. (1980) study (see Chapter 13), although they recorded no significant change in the mean tracheal mucociliary transport rates.

Clearly, the results of the donkey studies support the human experiments (Chapter 13) that indicate that sub-micrometer H_2SO_4 aerosols affect mucociliary clearance in the distal conductive airways. Mucociliary clearance depends on both the physicochemical properties of the mucus and the coordinated beat of the underlying cilia. Mucus is excreted into the airway lumen in an alkaline form that is then acidified by CO_2 (Holma et al., 1977). In vitro studies have shown that mucus is a solution in high pH solutions, while at lower pH it becomes viscous (Breuninger, 1964). The H^+ supplied by the H_2SO_4 may stiffen the mucus and increase the efficiency of removal. This is consistent with the increase in bronchial clearance rate observed in humans following exposure to 0.1 mg/m^3 H_2SO_4. Major changes in mucous viscosity could also impair clearance by making the mucus so stiff that ciliary movement is not possible. Other studies (Grose et al., 1980; Schiff et al., 1979) have shown that exposures to 0.9 to 1.1 mg/m^3 H_2SO_4 can cause a depression of tracheal ciliary beat frequency in hamsters that may lead to a depression in overall bronchial clearance. See Sections 12.4.1.1 and 12.4.2 for more details on these latter studies (Grose et al., 1980; Schiff et al., 1979) that were conducted with pollutant mixtures.

Based on the results summarized above, it is possible that chronic H_2SO_4 exposures at concentrations of about 0.1 mg/m^3 could produce persistent changes in mucociliary clearance (see Table 12-9).

Cadmium and nickel chlorides also disrupt the activity of the ciliated epithelium (Adalis et al., 1977, 1978). Tracheal rings have been isolated from hamsters and the beat frequency and morphology of the ciliated epithelium have been observed. Concentrations of $CdCl_2$ as low as 6 µM in vitro resulted in decreased beat frequency and degradation of the ciliated epithelium architecture (Adalis et al., 1977). A prior 2-h exposure in vivo to 2 µm aerosols of $CdCl_2$ at 0.05 to 1.42 mg/m^3 caused a significant decrease in cilia beat frequency proportional to the aerosol concentration. When hamsters were exposed to 1.33 mg/m^3 Cd^{++} for 2 h/day for 2 days, the beat frequency did not return to control values until 6 wk after exposure. Nickel chloride aerosols or solutions had similar, but less marked, effects (Adalis et al., 1978). The beat frequency descreased by 60 beats/min on exposure to 0.1 mg/m^3 Ni^{++} for 2 hours. The decline in beat frequency was proportional to the concentration of Ni^{++} aerosol or solution. A single 2-h exposure to 0.1 mg/m^3 Ni^{++} depressed cilia beat frequency

TABLE 12-9. EFFECTS OF H_2SO_4 ON MUCOCILIARY CLEARANCE

Concentration	Duration	Species	Results	Reference
0.1 mg/m^3 H_2SO_4	1 h/day, 5 day wk, six mo	Donkey	All four animals developed erractic bronchial mucociliary clearance rates within the first few weeks. Those animals never preexposed before the 0.1 mg/m^3 H_2SO_4 had slowed clearance during the second 3 mo of exposure.	Schlesinger et al., 1979
0.19 to 1.4 mg/m^3 H_2SO_4 (0.3 to 0.6 µm, MMAD)	1 h	Donkey	Bronchial mucociliary clearance was slowed	Schlesinger et al., 1978
0.5 mg/m^3 H_2SO_4	1 h	Dog	Slight increases in tracheal mucociliary transport velocities immediately and 1 day after exposure. One wk later clearance was significantly decreased.	Wolff et al., 1979
1.0 mg/m^3 H_2SO_4 (0.9 µm, MMAD, σ_g 1.4)	1 h	Dog	Depression in tracheal mucociliary transport rate persisted at 1 wk postexposure	Wolff et al., 1979
1.4 mg/m^3 H_2SO_4 (0.3 to 0.6 µm, MMAD)	1 h	Donkey	No effect on tracheal transport	Schlesinger et al., 1978
1.5 mg/m^3 H_2SO_4 (0.6 µm, CMD)	90 min	Mouse	No significant effects	Fairchild et al., 1975a
14 mg/m^3 H_2SO_4 (0.12 µm MMAD)	Short-term	Sheep	No significant changes in tracheal mucociliary transport rate	Sackner et al., 1978a
15 mg/m^3 H_2SO_4 (3.2 µm, CMD)	4 h	Mouse	Exposure to H_2SO_4 after exposure to a nonviable streptococcal aerosol reduced the rate of ciliary clearance from the lungs and nose	Fairchild et al., 1975a
15 mg/m^3 H_2SO_4 (3.2 µm, CMD)	90 min	Mouse	Exposure to H_2SO_4 4 days prior to bacterial aerosol. Clearance of nonviable bacteria reduced in nose, but not lungs.	Fairchild et al., 1975a

24 h after exposure, but the frequency returned to near normal values after 72 hours. After exposure to 0.1 mg/m^3, Cd^{++} was about 20 percent more effective than Ni^{++} in slowing cilia beat (Table 12-10).

12.3.4.2 Alveolar Macrophages--The cytotoxicity of components of atmospheric aerosols has been studied with alveolar macrophages (AM). The physiological role of AM in the prevention of infection and the defense of the lung through removal of inhaled particles has been amply demonstrated (Green, 1970).

The viability of guinea pig alveolar macrophages was decreased by Min-u-sil silica (6.8, 4.5, and 2.7 μm MVD), with the effect increasing as particle size decreased (Ottery and Gormley, 1978).

Aranyi et al. (1979) reported cytotoxic effects on AM with fly ash particles coated with PbO, NiO, or MnO_2. The percentage of metal adsorbed on the fly ash was fairly similar across particle size for a given metal. The fly ash particles were of three size ranges: <2, 2-5, or 5-8 μm in diameter. All of the particles, regardless of the coating or particle size, decreased cell viability and were phagocytized by the AM. Within a given chemical series of coated particles, the effects were both concentration- and size-related, with smaller particles and greater concentrations producing greater effects. It was suggested that the greater surface area of the smaller particles was responsible for the greater toxicity of the small particles. Total cellular protein and lactate dehydrogenase (LDH) also decreased after treatment, probably as a nonspecific result of the death of the cultured AM. For each particle size, Pb-coated particles were most toxic, NiO- and MnO_2-coated particles had intermediate effects, and the untreated fly ash was least toxic. The toxicity did not appear related to the solubility of the metal oxide coating, since no soluble metal could be found using the AM themselves as a bioassay. The toxicity appeared to be associated with the uptake of the intact particle. No changes were observed in the total lysosomal enzyme content, but the latency or intactness of the lysosomal membrane was not examined. Toxicity could have resulted from the disruption of the intracellular lysosomal membrane, which in turn could have released intracellular lysosomal enzymes. Lysosomal enzyme release has been proposed as one potential mechanism for the toxicity of asbestos and silica particles (Heppleston, 1962). These results support the concept that the surface activity of particles determines the toxicity of the particle (Allison and Morgan, 1979).

Camner et al. (1974) exposed rabbit alveolar macrophages *in vitro* to 5 μm Teflon particles coated with Al, Be, C, Pb, Mn, Ag, and U. All particles were phagocytized by the cells, but only Be caused a decrease in viability.

TABLE 12-10. EFFECTS OF METALS AND OTHER PARTICLES ON HOST DEFENSE MECHANISMS

Concentration	Duration	Species	Results	Reference
0.01 or 0.15 mg/m^3 Pb_2O_3 (0.18 μm, MMAD)	3 mo	Rat	Decreased the number of alveolar macrophages/lung	Bingham et al., 1968
0.01 mg/m^3 (0.17 μm, MMAD) $PbCl_2$ or 0.11 mg/m^3 (0.32 μm, MMAD) $NiCl_2$ or 0.15 mg/m^3 (0.15 μm, MMAD) Pb_2O_3 or 0.12 mg/m^3 (0.17 μm, MMAD) NiO	12 h/day, 6 day/wk, 2 mo with $PbCl_2$, $NiCl_2$, or NiO; continuously for 2 mo with Pb_2O_3	Rat	Exposure to Pb_2O_3 resulted in a depression of the number of alveolar macrophages (AM) for up to 3 mo but returned to control levels within 3 days after discontinuation. NiO produced a marked AM elevation. $NiCl_2$ resulted in marked increases in mucus secretion and bronchial hyperplasia.	Bingham et al., 1972
0.05 to 1.42 mg/m^3 $CdCl_2$	2 h	Hamster	Decreased ciliary beating frequency in trachea	Adalis et al., 1977
0.1 mg/m^3 $NiCl_2$	2 h	Hamster	Decreased ciliary beating frequency in trachea	Adalis et al., 1978
Graded concentrations: 0.075 to 1.94 mgCd/m^3 as $CdCl_2$ 0.1 to 0.67 mgNi/m^3 as $NiCl_2$, or 0.5 to 5 mgMn/m^3 as Mn_3O_4; all aerosols (94-99%) <1.4 μm in diameter	2 h	Mouse	The aerosols increased the mortality from the subsequent standard airborne streptococcal infection: $CdCl_2$ affect the response at 0.1 mg/m^3 $NiCl_2$ at 0.5 mg/m^3 and Mn_3O_4 at 1.55 mg/m^3.	Gardner et al., 1977b Adkins et al., 1979, 1980c
109 mg/m^3 MnO_2 (0.70 μm, mean diameter)	3 h/day	Mouse	Increased mortality after 3 or 4 days exposure when mice received bacterial aerosol immediately after exposure. When the bacteria were administered 5 hr post pollutant exposure, a single 3 hour exposure increased mortality. In mice exposed to aerosols of virus 1 or 2 days prior to MnO_2, there were also increased mortality and pulmonary viral lesions.	Maigetter et al., 1976
0.2 mg/m^3 $CdSO_4$, 0.6 mg/m^3 $CuSO_4$, 1.5 mg/m^3 $ZnSO_4$, 2.2 mg/m^3 $Al_2(SO_4)_3$, or 3.6 mg/m^3 $MgSO_4$	3 h	Mouse	Estimated concentrations that caused a 20% enhancement of bacterial-induced mortality over controls	Ehrlich et al., 1978 Ehrlich, 1979

TABLE 12-10. (continued)

Concentration	Duration	Species	Results	Reference
Ammonium sulfate at 5.3 mg/m^3 SO_4, NH_4HSO_4, at 6.7 mg/m^3 SO_4, NO_2SO_4 at 4 mg/m^3 SO_4, $Fe_2(SO_4)_2$ at 2.9 mg/m^3 SO_4, or $Fe(NH_4)_2SO_4$ at 2.5 mg/m^3 SO_4	3 h	Mouse	No significant alterations of host defense mechanisms	Ehrlich et al., 1978 Ehrlich, 1979
5.0 mg/m^3 carbon black or 2.5 mg/m^3 iron oxide	2 h	Mouse	No significant increases in mortality resulted upon subsequent exposure to airborne infection	Gardner, 1981
0.19 mg/m^3 $CdCl_2$ 0.25 mg/m^3 $NiCl_2$	2 h	Mouse	Decreased number of antibody-producing spleen cells	Graham et al., 1978

Although White and Kuhn (1980) did not consider particle size, they did conduct in vitro AM studies with iron carbonyl, SiO_2 (size not given), crocidolite and crysotile asbestos (size not given), kaolinite (size not given), and polystyrene latex beads (1.1 μm diameter). Although particle to cell ratios were roughly equivalent (10 to 15 particles/ cell), the particle concentration differed markedly for each chemical. Enzyme release was measured. Compared to a no particle control group, iron carbonyl, SiO_2, both forms of asbestos and latex beads, but not kaolinite, increased ($p < 0.02$) the percent of extracellular β-glucuronidase (a lysosomal enzyme). Similar results were obtained for percent extracellular LDH, except latex beads had no significant effect. All particles, except crocidolite asbestos and latex beads, increased elastase secretion.

Allison and Morgan (1979) have summarized the evidence that AM ingest both toxic and nontoxic particles in the same manner. In the case of fibers, ingestion appears more dependent upon the length of the fiber (Allison, 1973). Short fibers >5 μm are almost always ingested, while fibers >30 μm are seldom ingested completely and remain in contact with the plasma membrane as well as with the lysosomal surface. Intermediate sized particles (5 to 20 μm) are sometimes completely ingested and sometimes not. Once ingested, particles have two effects. An immediate cytotoxicity appears that is apparently due to the interaction of the particle with the plasma membrane (Allison and Morgan, 1979). This interaction is similar to the hemolytic effects described for silica particles. The second effect results in delayed cytotoxicity and occurs after the particle has been ingested into a primary phagocytic vacuole that then combines with a primary lysosome to yield a secondary lysosome containing the particle (Allison and Morgan, 1979). Here toxic particles exert an effect upon the permeability of the lysosomal membrane, resulting in the release of lysosomal enzymes into the cell and into the external medium. These proteolytic enzymes can cause tissue damage.

Hatch et al. (1980) examined the influence of in vitro exposure to a variety of particles on AM oxidant production (O_2^- and H_2O_2) and found the response to be chemical specific. All the particles studied stimulated chemiluminescence, with amphibole asbestos the most active. Silica, chrysotile asbestos, and metal oxide (Pb, Ni, Mn)-coated fly ash had intermediate activity. Fugitive dusts and fly ash had the lowest activity.

Waters et al. (1974) found that AM cultured with particulate forms of vanadium had decreased cell viability, indicating a direct cytotoxicity. Alveolar macrophages were cultured in medium containing vanadium pentoxide (V_2O_5), vanadium trioxide (V_2O_3), or vanadium dioxide (VO_2). Cytotoxicity was directly proportional to the solubility of the vanadium compound: $V_2O_5 > V_2O_3 > VO_2$. The concentration of V required to produce a 50-percent decrease in viability after 20 h of culture was found to be: 13 μg V/ml as V_2O_5, 21 μg V/ml as V_2O_3, and 33 μg V/ml as VO_2. When V_2O_5 was dissolved in the medium prior to incubation with the AM, only about 9 μg V/ml were required to reduce viability by 50 percent, thus indicating that the soluble V was responsible for toxicity. Phagocytosis, an essential function for the defense of the lung, was decreased by 50 percent with 6 μg V/ml as dissolved V_2O_5. Acid phosphatase, a lysosomal degradation enzyme necessary for digestion of phagocytized bacteria, was inhibited by 1 μg V/ml as V_2O_5, while the lysosomal enzymes, lysozyme and β-glucuronidase, were not inhibited by concentrations as high as 50 μg V/ml.

The effects of Fe_2O_3 on AM have also been investigated. Rabbits were exposed for 3 h to 186 to 222 mg/m^3 Fe_2O_3 (0.17-0.31 μm, MMAD), and AM were removed 0, 12, 18, and 24 h later (Grant et al., 1979). When selected lysosomal enzyme activites were determined for the first three postexposure times, there were no significant differences from control. However, since an increased ($p < 0.02$) number of cells were recovered after 12, 18, and 24 h postexposure, the total amount of some of the lysosomal enzymes in the lung was increased. It appears that the increased number of cells was due to the influx of smaller cells into the lung.

Alveolar macrophages exposed in vitro for 20 h to metallic salts were also studied by Graham et al. (1975b) using a technique to determine phagocytosis of viable cells only. The chlorides of Cd^{++} (2.2×10^{-5}M), Cr^{++} (3.1×10^{-3}M), Mn^{++} (1.8×10^{-3}M), and Ni^{++} (5.1×10^{-4}M) significantly inhibited phagocytosis. Ammonium vanadate (6.9×10^{-4}M) had no effect on phagocytosis, but did lyse and kill cells. Nickel, which caused the greatest reduction in phagocytosis, had very little effect on viability or cell lysis Antibody-mediated rosette formation of AM was also inhibited in vitro by low concentrations of $CdCl_2$ (2.2×10^{-5}M) or $NiCl_2$ (10^{-4}M) (Hadley et al., 1977). Inhibition was proportional to the Ni^{++} or Cd^{++} concentration and reached its maximum within 20 min. These studies showed that the antibody-dependent recognition system of AM was inhibited by trace concentrations of Ni^{++} and Cd^{++} almost immediately after contact with these metal ions. Such an effect implies that these metals may affect receptors for phagocytosis of opsonized bacteria. Depression of AM viability, phagocytosis, and receptors for phagocytosis may be a mechanism by which these heavy metal salts increase the susceptibility to airborne infections as discussed later (Section 12.3.4.3).

Bingham and coworkers (1968, 1972) examined the effects of Pb^{++} and Ni^{++} inhalation on the number and type of AM present in the lungs of rats. In a preliminary report, Bingham et al. (1968) showed that a 3-mo exposure to 0.01 or 0.15 mg/m^3 Pb_2O_3 (0.18 μm, MMAD) decreased the number of AM/lung. The specificity of this response was investigated in a subsequent study (Bingham et al. 1972) using soluble $PbCl_2$ (0.1 mg/m^3, 0.17 μm MMD) and $NiCl_2$ (0.11 mg/m^3, 0.32 μm MMD) and insoluble Pb_2O_3 (0.15 mg/m^3, 0.15 μm MMD) and NiO (0.12 mg/m^3, 0.25 μm MMD) aerosols. Rats were exposed for 12 h/day, 6 days/wk for 2 months. The only exceptions were those exposed to Pb_2O_3 continuously. Exposure to Pb_2O_3, but not $PbCl_2$, aerosols resulted in a depression of the number of AM that persisted throughout the experiment. The number of AM was depressed on inhalation of 0.15 mg/m^3 Pb_2O_3 for up to 3 mo, but returned to control levels within 3 days after discontinuing the exposure. The solubility of the Ni^{++} compound also effected the biological response. Nickel oxide produced a marked elevation in the number of AM/lung, while $NiCl_2$ did not. The most significant effects in $NiCl_2$-exposed rats were marked increases in mucus secretion and bronchial hyperplasia. No morphological alterations were observed in those rats exposed to $PbCl_2$ or Pb_2O_3. Isolated AM also varied in diameter with the exposure, but the biological significance of this size variation is not presently known. Perhaps different cell populations were recruited into the lung with the differing exposure conditions.

Cadmium chloride aerosols also altered the number and kind of cells recoverable by lavage following exposure (Gardner, 1977b, 1981). The total number of AM isolated from exposed rats decreased following exposure to 1.5 mg/m^3 Cd^{++} (99 percent <3 µm in diameter) but returned to normal values within 24 hours. The viability of the isolated cells decreased by 11.2 percent immediately after exposure and was still depressed 24 h later. There was an influx of polymorphonuclear leukocytes, especially 24 h postexposure, but no increase in lymphocytes. These effects were not observed at 0.5 mg/m^3 Cd^{++}, indicating that the minimum effective dose may lie somewhere between these two concentrations.

Nickel chloride aerosols (Adkins et al., 1979; Gardner 1981) produced neither an effect on the number of AM isolated by lavage of rats the day following a 2-h exposure to 0.65 mg/m^3 Ni^{++} nor an influx of polymorphonuclear leukocytes. The phagocytic capacity of the isolated AM was, however, depressed. A 2-h exposure of mice to 0.9 mg/m^3 Mn_3O_4 reduced the number of AM that could be recovered by lavage, but did not result in an influx of other cell types (Adkins et al., 1980a). The AM had a reduced concentration of ATP and total protein and acid phosphatase activity. Viability and phagocytic activity of AM were normal.

The number, function, and kind of cells isolated from the lung by lavage are influenced by the prior exposure to heavy-metal containing aerosols. Not all of these produced the same effect, but those containing Cd^{++}, Ni^{++}, and Mn^{++} also enhanced the susceptibility of mice to subsequent airborne infections (Gardner, 1981). The observations of two independent laboratories (Bingham et al., 1968, 1972; Adkins et al., 1979) on $NiCl_2$ aerosols are essentially in agreement (Table 12-10).

12.3.4.3 Interaction with Infectious Agents--Gardner (1981) and Ehrlich (1979) have reviewed their groups' studies and presented new data on the effects of aerosols on host defense mechanisms against infectious pulmonary disease in mice. In all of the Gardner studies, 94 to 99 percent of the aerosols was less than 1.4 µm in diameter (Gardner et al., 1977b; Gardner, 1981). Animals were placed in a head-only exposure system for 2 h and were given graded concentrations ranging from 0.075 to 1.94 mg/m^3 $CdCl_2$ (Gardner et al., 1977b), from 0.1 to 0.67 mg/m^3 $NiCl_2$ (Adkins et al., 1979), or from 0.5 to 5 mg/m^3 Mn_3O_4 (Adkins et al., 1980c). In mice, these exposures resulted in the deposition of 0.002 to 0.026 mg Cd^{++} (Gardner et al., 1977b), 0.001 to 0.012 mg Ni^{++} (Adkins et al., 1979), or 0.005 to 0.042 mg Mn^{++} (Adkins et al., 1980b) per g dry weight of lung, respectively. Nickel clearance (Graham et al., 1978) from the lungs of mice had a half-life of 3.4 days, while Mn^{++} (Adkins et al., 1980b) clearance was rapid, with a half-life of only 4.6 hours. None of the exposures appeared to be edemagenic, as judged by the ratio of dry weight to wet weight of the lung. After exposure, mice were challenged with an aerosol of Streptococcus pyogenes (S. pyogenes). The aerosols of $CdCl_2$ (Gardner et al., 1977b), $NiCl_2$ (Adkins et al., 1979), or $MnCl_2$ (Gardner, 1981) increased the mortality from the subsequent standard airborne infection. Cadmium was more toxic than Ni^{++}, which was more toxic than Mn^{++}. Exposure to Cd^{++} and Mn^{++} resulted in a significant linear concentration response. The lowest concentration tested at which a significant increase in mortality was detected was 0.1 mg/m^3 Cd or 0.5 mg/m^3 Ni^{++}. Manganese, as Mn_3O_4 (Adkins et al., 1980c), was statistically estimated to produce a 10-percent increase in mortality at 1.55 mg/m^3 Mn^{++}, while $MnCl_2$ (Gardner, 1981) required a higher concentration to produce a measurable increase in mortality. Using a different infectivity model (Maigetter et al., 1976), 3 or 4 days (3 h/day) of exposure to 109 mg/m^3 MnO_2 (0.70 μm, mean diameter) were required to increase mortality consequent to Klebsiella pneumoniae infection when the mice received the bacterial aerosol immediately after exposure.

The toxicity of $NiCl_2$ is complex (Adkins et al., 1979). Nickel exposure had no effect on the S. pyogenes infection if the bacteria were given immediately after Ni^{++} aerosol exposure. When the bacterial exposure was delayed by 24 h, Ni^{++} aerosols increased the mortality in a concentration-related fashion. In contrast, effects of $CdCl_2$ (Gardner et al., 1977b) and Mn^{++} (Gardner, 1981) were observed when the bacterial challenge immediately followed exposure. The concentration-response curve of Ni^{++} was very steep compared to those of Cd^{++} and Mn^{++} exposures (Gardner, 1981). No explanation has been offered for the delay in effect of Ni^{++}. Perhaps the delayed effects represent either redistribution of Ni^{++} to the site of action or some major change in the lung, such as death of a specific cell type. The delayed toxicity raises the possibility of carry-over of effects from a single exposure to a second.

The influence of a variety of sulfate species on host defense mechanisms against infectious respiratory disease has been investigated by Ehrlich (1979) and Ehrlich et al. (1978) using the infectivity model with S. pyogenes. Mice were exposed for 3 hours. The estimated concentrations of the compounds that caused a 20-percent enhancement of bacterial-induced mortality over controls were 0.2 mg/m^3 $CdSO_4$, 0.6 mg/m^3 $CuSO_4$, 1.5 mg/m^3 $ZnSO_4$, 2.2 mg/m^3 $Al_2(SO_4)_3$, 2.5 mg/m^3 $Zn(NH_4)_2(SO_4)_2$, and 3.6 mg/m^3 $MgSO_4$. Ammonium sulfate at 5.3 mg/m^3 SO_4, NH_4HSO_4 at 6.7 mg/m^3 SO_4, Na_2SO_4 at 4 mg/m^3 SO_4, $Fe_2(SO_4)_3$ at 2.9 mg/m^3 SO_4, and $Fe(NH_4)_2SO_4$ at 2.5 mg/m^3 SO_4 did not cause significant alterations. The nitrates of Pb^{++}, Ca^{++}, Na^+, K^+, and NH_4 did not cause an effect at concentrations of 2 mg/m^3 or higher; however, $Zn(NO_3)_2$ caused effects similar to $ZnSO_4$. From this research, it appears that the NH_4 ion rendered the compound less toxic, and that the toxicity is due primarily to the cation. With the infectivity model, $ZnSO_4$, $Zn(NH_4)_2(SO_4)_2$, and $CuSO_4$ ranked differently than with airway resistance experiments (Amdur et al., 1978a). This is not unexpected, as airway resistance primarily detects alterations of the medium to large conducting airways, while the infectivity model (Gardner and Graham, 1977) is hypothesized to reflect alveolar level changes.

When mice were exposed for 2 h to 5.0 mg/m^3 carbon black or 2.5 mg/m^3 iron oxide, no significant increases in mortality resulted on subsequent exposure to airborne infection (Gardner, 1981).

Death from S. pyogenes exposure in this infectivity model is due to septicemia (Gardner et al., 1977b). Septicemia occurs when the bacteria have grown to 10^5 organisms per lung. Removal and killing of the inhaled organisms reduces the growth of the bacteria within the host and prevents the occurrence of septicemia. For these reasons, the infectivity model is an integrative assessment of toxicity for host defense systems against infectious pulmonary disease. As reported above, the number, kind, function, and viability of the cells isolated by lavage from the lungs of animals exposed to heavy-metal containing aerosols are different from those of control animals. Studies of tracheal rings isolated from aerosol-exposed hamsters also indicate depression of mucociliary clearance. Both mucocilary and AM clearance of bacteria are depressed by aerosols of these heavy metals (Gardner, 1981) (Table 12-10).

12.3.4.4 Immune Suppression--Antibodies play a significant role in the ability of macrophages to recognize and engulf pathogenic bacteria. The functioning of the immune system also interlocks with the macrophage system in other ways. In mice, intramuscular injections of $NiCl_2$ depressed the number of antibody-producing cells in the spleen (Graham et al., 1975a). Using the Jerne plaque assay, a negative linear dose-response curve was found with injections ranging from 0.093 to 0.123 mg Ni^{++}/g body weight. No effect was observed with a dose of 3.09 µg Ni^{++}/g body weight. Graham et al. (1978) calculated that exposure to an aerosol of 0.25 mg/m^3 Ni^{++} for 2 h results in a maximum deposition of 0.98/mg Ni^{++}, assuming complete retention and a minute volume of 1.45 ml/g body weight. This concentration was found to be the lowest tested that produced a significant depression in the immune response. The lowest dose found to produce a similar effect by injection was 0.21 mg Ni^{++}/mouse (Graham et al., 1975a). The inhalation dose was, therefore, approximately 200 times more potent. Nickel was found to follow first order removal kinetics from the lung, but measurable elevations remained in the lung up to 4 days after exposure. Similar kinetics of removal have been found using the isolated, ventilated, and perfused rat lung (Williams et al., 1980) and human, rat, and cat type II pneumocytes in culture (Saito and Menzel, 1978).

Inhaled Cd^{++} also depresses the number of antibody-producing cells and is more potent than intramuscularly injected Cd^{++}. The highest intramuscular dose of $CdCl_2$ examined by Graham et al. (1978) was 0.012 mg Cd^{++}/g body weight (about 0.266 mg Cd^{++}/mouse), and it produced no immunosuppression. When mice were exposed to 0.19 mg/m^3 Cd^{++} for 2 h, a significant suppression was observed. In both cases, the Cd^{++} was administered as $CdCl_2$, a highly soluble salt. The inhalation dose can be calculated, on the same basis as that given above for Ni^{++}, to be at most 0.0007 mg $CdCl_2$/mouse. The inhaled dose was, therefore, at least 350-fold more potent. Inhalation also appeared to be more potent than ingestion or interperitoneal injection (Exon et al., 1975; Koller et al., 1975). Koller et al. (1975) found that 0.150 mg Cd^{++} given orally was required to produce immunosuppression.

For comparative purposes, the lowest inhalation exposure of $CdCl_2$ found to be immunosuppressive was 0.19 mg/m^3; 0.2 mg/m^3 was the 1971 Threshold Limit Value (TLV). The current TLV is 0.05 mg/m^3. Conservative estimates of the human intake from air and water are 0.0074 mg/day and 0.160 mg/day, respectively (Schroeder, 1970). $NiCl_2$ was found to be immunosuppressive at an inhalation exposure of 0.25 mg/m^3, while its TLV is 1 mg/m^3. The human exposure is estimated to be 0.0024 mg/day from inhalation and 0.60 mg/day from ingestion (Schroeder, 1970). Should the effectiveness of inhaled aerosols be equivalent in mice and men, then the inhaled doses are biologically almost equivalent to those ingested.

Inhaled Cd^{++} or Ni^{++} aerosols impair the bacterial defenses of the lung through direct cytotoxicity to AM, inhibition of antibody dependent aggregation reactions, and perhaps by depression of antibody production. All of these mechanisms might help to explain the increased susceptibility of mice to airborne pathogens following inhalation of Ni^{++} or Cd^{++} aerosols. The rapidity of clearance of Ni^{++} and Cd^{++} from the lung may allow rapid recovery (see Table 12-10).

Mitogen-induced transformation (a reflection of immune function) has been measured in mouse splenic lymphocytes exposed _in vitro_ to 0.50 mg of various sizes of silica (Wirth et al., 1980). Four silica samples were tested unfractionated, or size-fractionated into two categories (0.3 and 5.3 μm). All the unfractionated samples depressed the blastogenic response to Conconavalin A (a measure of T cell function) and lipopolysaccharide (a measure of B cell function). The T cell response was decreased by the 0.3 μm size fraction of all samples. The 5.3 μm particles of 2 samples increased the response, however, while one sample caused no change and another caused a small decrease. When B cell function was examined, it was more depressed by the 0.3 μm silca than the 5.3 μm particles, although three of the four larger-sized samples did cause a decrease.

12.4 INTERACTION OF SULFUR DIOXIDE AND OTHER POLLUTANTS

12.4.1 Sulfur Dioxide and Particulate Matter

Although man breathes a multitude of chemicals in various mixtures at various dose-rates, most animal toxicological and controlled human exposures are conducted with single chemicals. This simplifies the research and permits an improved estimate of cause-effect relationships, but it prohibits evaluation of the effects of pollutant mixtures that may be additive, synergistic, or antagonistic with respect to the individual pollutants. Some interaction studies that elucidate the complexity of toxicological interrelationships, however, have been conducted. Some of this work used pollutant combinations that favored the conversion of the primary pollutant to a secondary pollutant (i.e., SO_2 altered to H_2SO_4, etc.). Other research was directed at evaluating the influence of several pollutants when delivered in combination or in sequence.

12.4.1.1 Acute Exposure Effects--The question of the possible effect of aerosols on the response to SO_2 is important in air pollution toxicology (Amdur, 1977). The phenomenon has been investigated in simple model systems of SO_2 alone or in combination with an aerosol of a single chemical. The typical bioassay system has been the comparison of the increase in pulmonary flow resistance in guinea pigs produced by a given concentration of SO_2 alone with that produced in the presence of the aerosol. The aerosols used in many of these studies were "inert" in the sense that they did not produce an alteration in flow resistance when they were given alone.

The initial simple prototype aerosol used was sodium chloride (NaCl) at concentrations of 10 mg/m^3 and 4 mg/m^3 (Amdur, 1961). These experiments with guinea pigs indicated that the response to a given concentration of SO_2 was potentiated by 10 mg/m^3 NaCl. For example, a concentration of 5.24 mg/m^3 (2 ppm) SO_2 alone produced an increase of 20 percent in pulmonary flow resistance; when the NaCl was present, the increase was 55 percent. The potentiation did not occur until the latter part of a 1 h exposure. When the concentration of sodium chloride was reduced to 4 mg/m^3, the potentiation was greatly reduced. Examination of postexposure data indicated that the response to the combination resembled the response to a more irritant aerosol. The length of recovery was related to the concentration of SO_2, and the presence of the aerosol delayed recovery to control values. The chamber relative humidities were below 70 percent; but on entering the high humidity of the respiratory tract, the NaCl would absorb water to become a droplet capable of dissolving SO_2, thus favoring the production of H_2SO_4. Sodium chloride alone does not catalyze the oxidation of SO_2 to H_2SO_4.

Experiments by McJilton et al. (1973, 1976) indicate the importance of ambient RH and the solubility of SO_2 in the sodium chloride droplet. They examined the effect of 1 mg/m^3 NaCl on the response to 2.62 mg/m^3 (1 ppm) SO_2 at low (<40 percent) and high (>80 percent) RH. An increase in pulmonary flow resistance in guinea pigs was the response criterion. As would have been predicted from the earlier work, no increase was observed with this NaCl concentration at low RH. At high RH, the potentiation was marked and evident during both the early and late parts of the 1-h exposure. The rapid onset indicates the formation of an irritant aerosol in the exposure chamber under high humidity. Predictably, no conversion to sulfate was found, but the droplets were acidic with an estimated pH of 4. Presumably, this was sulfurous acid. (See the discussion of the effect of RH on sulfate and nitrate aerosols and the interaction of NaCl above and on human exposure experiments in Chapter 13).

Using the Mead-Amdur method, Amdur and Underhill (1968) studied the effect of aerosols of soluble salts of metals shown to convert SO_2 to H_2SO_4. Manganous chloride, ferrous sulfate, and sodium orthovanadate caused a three-fold increase in the resistance to flow over that of 2.62 mg/m^3 (1 ppm) SO_2 alone. The potentiation was evident during the first 10 min as well as during the remainder of the 1-h exposure. Chamber RH was 50 percent, indicating that high humidity was not necessary for the formation of an irritant aerosol in the chamber when the catalyzing metals were present. Analysis of the collected aerosol indicated the presence of sulfate, presumably as H_2SO_4 (Amdur, 1973). These analyses indicated that, at an SO_2 concentration of 0.52 mg/m^3 (0.2 ppm), about 0.08 mg H_2SO_4 was formed. When this amount of H_2SO_4 was administered with 0.52 mg/m^3 (0.2 ppm) SO_2, the increase in flow resistance duplicated the increase observed with the iron and vanadium aerosols (Amdur, 1974). This suggests that H_2SO_4 formation is a likely mechanism of potentiation for the aerosols of these metals, although the formation of stable sulfite complexes in the air may also account for this effect (Hansen et al., 1974; Schlesinger et al., 1980).

Amdur et al. (1978a) have reported that a 1-h exposure to 0.4 mg/m^3 copper sulfate also potentiated the response to 0.94 mg/m^3 (0.36 ppm) SO_2. It is not certain whether this is caused through the formation of H_2SO_4 or the formation of a sulfite complex. The increased resistance to flow from exposure to 0.79 to 0.84 mg/m^3 (0.3 to 0.32 ppm) SO_2 with ammonium sulfate (0.9 mg/m^3), ammonium bisulfate (0.9 mg/m^3), or sodium sulfate (0.9 mg/m^3) was purely additive. It should be pointed out that these salts have not been tested under conditions of high relative humidity.

Amdur and Underhill (1968) also examined the effect of a variety of solid aerosols that do not catalyze the conversion of SO_2 to H_2SO_4. None of these potentiated the increased resistance to flow when compared to SO_2 (Table 12-11).

12.4.1.2 Chronic Exposure Effects--Animals were exposed continuously to various combinations of SO_2, H_2SO_4 (0.5 to 3.4 μm, MMD), and fly ash (3.5 to 5.9 μm, MMD) (Alarie et al., 1975). The fly ash had been collected downstream from electrostatic precipitators of coal-burning electric generating plants. Monkeys were exposed for 18 mo and guinea pigs for 12 months. The monkeys were exposed to SO_2, H_2SO_4 + fly ash, SO_2 + H_2SO_4, or SO_2 + H_2SO_4 + fly ash. Guinea pigs received either 0.9 mg/m^3 H_2SO_4 (0.49 μm MMD) or 0.08 mg/m^3 H_2SO_4 (0.54 or 2.23 μm MMD) + 0.45 mg/m^3 fly ash (3.5 or 5.31 μm MMD). In monkeys, a battery of hematological and pulmonary function (tidal volume, respiratory rate, minute volume, dynamic compliance, pulmonary flow resistance, work of breathing, distribution of ventilation, CO diffusing capacity, and arterial blood gases) tests were applied at various times during exposure, but no significant effects were attributed to the exposures. Similar methods (except for distribution of ventilation and CO diffusing capacity) were used with guinea pigs, and again no significant effects were observed. At the end of the exposure to 2.59 mg/m^3 (0.99 ppm) SO_2 + 0.93 mg/m^3 H_2SO_4 (0.5 μm MMD, σ_g = 1.5 to 3.8), the lungs of monkeys had morphological alterations in the bronchial mucosa (focal goblet cell hypertrophy and occasional hyperplasia and focal squamous metaplasia). Monkeys exposed to 2.65 mg/m^3 (1.01 ppm) SO_2 + 0.88 mg/m^3 H_2SO_4 (0.54 μm MMD, σ_g = 1.5 to 3.8) + 0.41 mg/m^3 fly ash (4.1 μm MMD, σ_g = 1.8 to 2.8) had similar alterations. Thus, fly ash did not enhance the effect. Monkeys that received 0.99 mg/m^3 H_2SO_4 (0.64 μm MMD, σ_g = 1.5 to 3.0) + 0.55 mg/m^3 fly ash (5.34 μm MMD, σ_g = 1.8 to 2.2) had slight alterations in the mucosa of the bronchi and respiratory bronchioles. Focal areas of erosion and epithelial hypertrophy and hyperplasia were observed. The other groups of monkeys had no remarkable morphological changes. No monkeys exposed to fly ash displayed morphological alterations, although the presence of the fly ash was easily observed. Guinea pigs had no morphological effects that could be attributed to pollutant exposure.

TABLE 12-11. EFFECTS OF ACUTE EXPOSURE TO SO_2 IN COMBINATION WITH CERTAIN PARTICLES

Concentration	Duration	Species	Results	Reference
5.24 mg/m^3 (2 ppm) SO_2, 10 mg/m^3 and 4 mg/m^3 NaCl	1 h	Guinea pig	5.24 mg/m^3 SO_2 alone produced an increase of 20% in pulmonary flow resistance; with NaCl at 10 mg/m^3 the increase was 55%.	Amdur, 1961
2.62 mg/m^3 (1 ppm) SO_2, 1 mg/m^3 NaCl at low (40%) and high (80%) (RH)	1 h	Guinea pig	No increase in pulmonary flow resistance at low RH. At high RH, the potentiation was marked.	McJilton et al., 1973
2.62 mg/m^3 (1 ppm) SO_2, an aerosol of soluble salts (manganous chloride, ferrous sulfate, and sodium orthovanadate) 50% RH	1 h	Guinea pig	Presence of soluble salt increased pulmonary flow resistance about 3-fold	Amdur and Underhill, 1968
0.94 mg/m^3 (0.36 ppm) SO_2, 0.4 mg/m^3 copper sulfate	1 h	Guinea pig	Potentiated pulmonary flow resistance	Amdur et al., 1978a
0.79 to 0.84 mg/m^3 (0.3 to 0.32 ppm) SO_2 and 0.9 mg/m^3 ammonium bisulfate, or 0.9 mg/m^3 sodium sulfate	1 h	Guinea pig	The effect on pulmonary flow resistance was additive	Amdur et al., 1978a

1 ppm SO_2 = 2.62 mg/m^3.

In a previous study, Alarie et al. (1973bc) found no effects on pulmonary function, hematology, or morphology of monkeys or guinea pigs exposed to approximately 0.56 mg/m^3 fly ash in combination with 3 concentrations of SO_2 (0.28, 2.62, or 13.1 mg/m^3; 0.11, 1, or 5 ppm). Monkeys were exposed continuously for 78 wk and guinea pigs continuously for 52 weeks.

Lewis et al. (1969, 1973) investigated the effects of SO_2 and H_2SO_4 in normal dogs and in dogs that had been exposed previously for 191 days to 48.9 mg/m^3 (26 ppm) NO_2. Dogs identically treated with NO_2 had morphological changes in the lung, and one of the animals had striking bullous emphysema. Sulfur oxide exposures were for 21 h/day for a maximum of 620 days to 13.4 mg/m^3 (5.1 ppm) SO_2, to 0.89 mg/m^3 H_2SO_4 (90 percent < 0.5 µm in diameter), or to a combination of the two. These concentrations were averaged over time, and when the animals were examined at 225 days, the concentration of H_2SO_4 was lower (0.76 mg/m^3 H_2SO_4 in the H_2SO_4 group and 0.84 mg/m^3 H_2SO_4 in the H_2SO_4 + SO_2 group). After 225 days of exposure (Lewis et al., 1969), dogs receiving H_2SO_4 had a significantly lower diffusing capacity for CO than those that did not receive H_2SO_4. In the SO_2-exposed animals, pulmonary compliance was reduced ($p < 0.05$), and pulmonary resistance was increased ($p < 0.05$) compared to animals that did not receive SO_2. Dogs not preexposed to NO_2 that received SO_2 + H_2SO_4 had a smaller residual volume ($p < 0.01$) than all other dogs.

These dogs were also examined after 620 days of exposure (Lewis et al., 1973). At 3, 7, 19, or 20.5 mo of exposure, SO_x did not markedly affect hematological indices. No morphological changes were clearly identified as resulting from SO_x exposure; however, pulmonary function was altered. Generally, the animals preexposed to NO_2 were more resistant to the SO_x. Sulfur dioxide did not produce any significant effects except for an increase in nitrogen washout. Sulfuric acid caused a significant ($p < 0.05$) decrease in diffusing capacity for CO, residual volume, and net lung volume (inflated) with an increase in total expiratory resistance. There was also a marginally significant ($p = 0.1$) decrease in total lung capacity, inspiratory capacity, and functional residual capacity. Total lung weight and heart weight also decreased. Other measurements (other lung volumes, dynamic and static compliance, and nitrogen washout) were not significantly affected. These alterations of diffusing capacity for CO and lung volumes are interpreted as a loss of functional parenchyma, and, along with the increase in total pulmonary resistance, are in the direction expected for animals that develop obstructive pulmonary effects. Although the standard histological techniques used did not detect morphological effects, it is conceivable that the pulmonary function effects preceeded measurable structural alterations.

Female beagle dogs were exposed 16 h/day for 68 mo to raw or photochemically-reacted auto exhaust, oxides of sulfur or nitrogen, or their combinations. Table 12-12 describes the exposure groups. More than 90 percent of the particles were <0.5 µm in diameter. The dogs were examined after 18 (Vaughn et al., 1969), 36 (Lewis et al., 1974), and 61 mo (Lewis et al.,

1974) of exposure and 24 mo (Gillespie, 1980) or 32-36 mo (Hyde et al., 1978; Orthoefer et al., 1976) after the 68 mo exposure ceased. A monograph describing the entire study and results is available (Stara et al., 1980). Only those results pertaining to SO_x are described here.

Although cardiovascular function was also assessed after 4 yr of exposure and 3 yr after exposure ceased, no significant changes attributable to SO_x were found (Gillespie, 1980). Typical hematological examinations (except for differential counts) were made approximately every 6 mo (Orthoefer et al., 1976). The SO_x group (see Table 12-12 for abbreviations) had no major differences from control. In the presence of auto exhaust (with or without irradiation), however, SO_x did cause some significant elevations in hematocrit and hemoglobin concentration. Clinical chemistries were unchanged during or approximately 1 1/2 yr after exposure (Gillespie, 1980).

A variety of other parameters were examined during or immediately after exposure (Gillespie, 1980). Sulfur oxides caused no significant effect on visual evoked brain potentials.

After 18 (Vaughn et al., 1969) or 36 mo (Lewis et al., 1974) of exposure, no significant changes in pulmonary function were observed. A variety of alterations were found using analysis of variance after 61 mo (Lewis et al., 1974) of exposure. Residual volumes were increased in dogs receiving R + SO_x compared to those receiving I + SO_x, SO_x, and clean air (CA). Residual volumes of the SO_x group were lower than those of the CA group. When χ^2 analyses were applied to the data of the number of dogs/group having alterations as judged by clinical criteria, additional significant differences were found. More dogs of the I + SO_x group had higher total expiratory resistance than their controls (CA and SO_x). The ratio of residual volume to total lung capacity was higher in animals exposed to R + SO_x, compared to those receiving CA. This change was interpreted as pulmonary hyperinflation. Although other lung volumes, compliance, resistance, diffusing capacity for CO, nitrogen washout, peak expiratory flow, and maximum breathing capacity were also measured, SO_x had no effects.

Two years after exposure ceased, pulmonary function was remeasured (Gillespie, 1980). These measurements were made in a different laboratory than those made during exposure, but consistency among measurements of the control group and another set of dogs of similar age at the new laboratory indicated that this difference did not have a major impact on the findings. Animals in the R, R + SO_x, and I + SO_x groups had an increased arterial pressure of CO_2 ($PaCO_2$) compared to controls ($p < 0.05$). These groups and the SO_x group had a greater dead space volume compared to controls. Respiratory frequency was increased in the SO_x group. Although the diffusing capacity of CO in the lung (DL_{CO}) was unchanged, the ratio of DL_{CO} to total lung capacity decreased in all pollutant-exposed dogs. Vital capacity did not change.

TABLE 12-12. POLLUTANT CONCENTRATIONS FOR CHRONIC EXPOSURE OF DOGS[a]

Atmosphere	Pollutant Concentration, mg/m^3						
	CO	HC (as CH_4)	NO_2	NO	OX (as O_3)	SO_2	H_2SO_4
Control Air (CA)[b]	-	-	-	-	-	-	-
Nonirradiated auto exhaust (R)	112.1	18.0	0.09	1.78	-	-	-
Irradiated auto exhaust (I)	108.6	15.6	1.77	0.23	0.39	-	-
$SO_2 + H_2SO_4 (SO_x)$[c]	-	-	-	-	-	1.10	0.09
Nonirradiated auto exhaust + SO_2 + H_2SO_4 (R + SO_x)	113.1	17.9	0.09	1.86	-	1.27	0.09
Irradiated auto exhaust + SO_2 + H_2SO_4 (I + SO_x)	109.0	15.6	1.68	0.23	0.39	1.10	0.11
Nitrogen oxides, 1 (NO_2 high)	-	-	1.21	0.31	-	-	-
Nitrogen oxides, 2 (NO high)	-	-	0.27	2.05	-	-	-

[a] Hyde et al., (1978).

[b] Abbreviations in parentheses

[c] >90% of H_2SO_4 particles were < 0.5 μm in diameter (optical sizing)

Compared to control values, total lung capacity and residual volume of the SO_x group were significantly increased; there were no changes in functional residual volume and respiratory, pulmonary, and chest wall resistance; and quasistatic chest wall compliance decreased. There was a greater change in dynamic compliance with increasing breathing frequency in dogs exposed to SO_x. When the pulmonary function values at the end of exposure were compared directly to those values 2 yr after exposure, the SO_x group had increases in residual volume, total lung capacity, vital capacity, inspiratory capacity, functional residual capacity, DL_{CO}, and the ratio of DL_{CO} to total lung capacity. The magnitudes of these changes were greater than changes in controls, in most cases. From evaluation of all the data, the authors indicated that functional loss continues following termination of the exposure, and the damage caused by SO_x is primarily to the parenchyma. They also stated that the combination of auto exhaust and SO_x "did not appear to augment specific functional losses caused by single species of pollutants."

Thirty-two to 36 mo (Hyde et al., 1978) after exposure, the lungs of the beagles were examined using morphologic (light, scanning electron and transmission electron microscopy) and morphometric techniques. In the SO_x group, lung weight, total lung capacity, and the displaced volume of the processed right lung were significantly increased over the controls (CA). In the most severely affected SO_x dogs, the air spaces enlarged and the number and size of interalveolar pores increased. Only the high-NO_2 group of dogs had a greater degree of air space enlargement. The SO_x-group animals had a loss of cilia in the conducting airways without squamous cell metaplasia, nonciliated bronchiolar cell hyperplasia, and loss of interalveolar septa in alveolar ducts. When SO_x was combined with R, cilia were also lost, but squamous cell metaplasia occurred. Exposure to R + SO_x and I + SO_x produced nonciliated bronchiolar cell hyperplasia and an increase in interalveolar pores and alveolar air space enlargement. The enlargement of the distal air spaces was centered on respiratory bronchioles and alveolar ducts and was associated with an apparent loss of interalveolar septa in all animals receiving SO_2 and H_2SO_4. The authors consider these changes analogous to an incipient stage of human proximal acinar (centrilobular) emphysema. The important observation from these experiments is that mixtures of SO_2 and H_2SO_4, representing an interacting gas-aerosol system similar to that in urban atmospheres, produced anatomic alterations at low concentrations.

In a monograph describing all the dog studies (Stara et al., 1980), the morphological and functional changes are compared. In the SO_x group, the changes in pulmonary function correlated well with the morphological effects. Since the changes in pulmonary function were progressive over the postexposure period, it is likely that morphological changes were also progressive.

Biochemical analyses were performed on these dogs at the time of sacrifice, 2.5 to 3 yr after exposure. Hydroxyproline concentration (used as an index of collagen content) and prolyl hydroxylase activity (the rate-limiting enzyme in collagen synthesis) were measured (Orthoefer et al., 1976). No significant changes in hydroxyproline were found. The SO_x and I + SO_x groups had significantly elevated prolyl hydroxylase activity compared to the R, R + SO_x, and CA groups. While it is remarkable that effects on prolyl hydroxylase remained 2.5 to 3 yr after exposure, it is not possible to interpret these results further. No significant alterations were observed in brain, heart, lung or liver lipids among the experimental groups (Gillespie, 1980).

Zarkower (1972) reported mixed effects on the immune system of mice exposed to 5.24 mg/m^3 (2 ppm) SO_2 and 0.56 mg/m^3 carbon (as carbon black, 1.8 to 2.2 μm, MMD), alone and in combination for 100 h/wk for up to 192 days. The animals were immunized with aerosols of bacteria (Escherichia coli) at various times during exposure. After 102 days of exposure, there were no statistically significant changes. Sulfur dioxide exposure caused an increase ($p < 0.05$) in serum antibody titer at 135 days and a decrease ($p < 0.01$) at 192 days. Carbon and SO_2 + carbon produced an equivalent decrease ($p < 0.01$) in antibody titer at 192 days (but not at 135 days) that appeared to be a greater decrease than that found in the SO_2-exposed mice. In the spleen, exposure to SO_2 caused an increase ($p < 0.01$) in the number of antibody-producing cells at 135 days and a decrease ($p < 0.01$) in number at 192 days. In the mediastinal lymph nodes (that drain the lung), SO_2 caused no such changes. Carbon + SO_2, but not carbon alone, caused an increase ($p < 0.01$) in the number of antibody-producing cells in the mediastinal lymph nodes and a decrease ($p > 0.05$) in the spleen at 135 days. After 192 days of exposure to carbon or carbon + SO_2, the number of antibody producing spleen cells decreased ($p < 0.01$). The immunosuppression in these 2 groups was roughly equivalent and appeared to be more severe than that in the SO_2 alone group. In the mediastinal lymph nodes, only carbon + SO_2 caused immunoenhancement ($p < 0.05$). Thus, for the pulmonary immune system, only exposure to the combination of SO_2 and carbon caused significant effects. After 192 days, the systemic immune system was affected in all 3 exposure groups. It appeared that carbon and carbon + SO_2 caused equivalent effects and that both regimens were more effective than SO_2 alone.

Fenters et al. (1979) showed that exposure for 3 h/day, 5 days/wk for up to 20 wk to a mixture of 1.4 mg/m^3 H_2SO_4 plus 1.5 mg/m^3 carbon (as carbon black, 0.4 μm, mean particle diameter) or to 1.5 mg/m^3 carbon only (0.3 μm, mean particle diameter) also altered the immune system of mice. Serum immunoglobulins (Ig) decreased, with the exception of IgM, which was increased after 1 wk of exposure to either carbon or H_2SO_4 + carbon. After 1 wk, some Ig classes decreased in both exposure groups, but after 4 or 12 wk of exposure, alterations were observed only in the H_2SO_4 + carbon group. Results for Ig were mixed at 20 weeks. In the carbon group, the number of specific antibody-producing spleen cells was increased at 4 wk, unchanged at 12 wk, and decreased at 20 weeks. A similar trend was observed in the H_2SO_4 + carbon group, but only the immunosuppression at 20 wk was significant. In examining other host defense systems, no alterations of AM viability or cell numbers were observed. After 4 and 12 wk of exposure, pulmonary bactericidal activity was increased in both exposure groups. By 20 wk of exposure, values were not significantly different from controls. Using the infectivity model with influenza A_2/Taiwan virus, a 20-, but not a 4-, wk exposure to H_2SO_4 + carbon increased mortality.

Morphological changes were observed in these mice (Fenters et al., 1979) using scanning electron microscopy after 12 wk of carbon exposure. In the external nares, there was excess sloughing of squamous cells. In the trachea, the number of mucous cells appeared to increase; dying cells were present, and microvilli were lost. No alterations of the bronchi were seen. The alveoli had some areas of congestion with thickening, loss of interalveolar septa, and enlarged pores. After 20 wk of exposure, damage was similar, but to a lesser degree. Mice exposed to the mixture of H_2SO_4 and carbon showed equivalent effects, but the damage was somewhat more severe than that seen in the carbon only group.

The influence of H_2SO_4 and carbon on the tracheas of hamsters was investigated by Schiff et al. (1979). Animals were exposed for 3 h to 1.1 mg/m^3 H_2SO_4 (0.12 μm, mean size) and/ or 1.5 mg/m^3 carbon (as carbon black, 0.3 μm, mean size) and were examined either immediately, or 24, 48, or 72 h later. Carbon caused no change in ciliary beat frequency. Sulfuric acid exposure, however, caused depression in this frequency at all times. The combination of H_2SO_4 and carbon produced similar effects, but recovery had occurred by 48 h postexposure. Using light microscopy, the percentage of normal tracheal epithelium was determined. Up to 48 h after exposure, the combination of H_2SO_4 and carbon resulted in more tissue destruction than either pollutant alone, although the single pollutants did cause some damage. Morphological alterations of all pollutant exposure groups were observed using light and scanning electron microscopy (see Table 12-13).

12.4.2 Interaction with Ozone

Cavender et al. (1977) exposed rats and guinea pigs to H_2SO_4 aerosols (10 mg/m^3, 1 μm MMD), 3.9 mg/m^3 (2 ppm) ozone (O_3), or a combination of the two for 6 h/day for 2 or 7 days; they then measured the ratio of lung to body weight and examined the lungs histologically. No synergism was observed between the O_3 and H_2SO_4 treatments. The histological lesions were those ascribed to O_3 alone. This same group (Cavender, 1978) exposed rats and guinea pigs to H_2SO_4 aerosols (10 mg/m^3, 0.83 μm, MMAD, σ_g = 1.66), 1.02 mg/m^3 (0.52 ppm) O_3, or a combination of the two for 6 h/day, 5 days/wk for 6 months. The histological alterations were those due to O_3 alone.

Last and Cross (1978) found synergistic effects of a continuous exposure of H_2SO_4 aerosol (1 mg/m^3) and O_3 (0.78 to 0.98 mg/m^3 or 0.4 to 0.5 ppm) when administered simultaneously to rats for 3 days. Glycoprotein synthesis was stimulated in tracheal ring explants measured *ex vivo*. Ozone alone caused a decreased glycoprotein secretion; H_2SO_4 was relatively inactive,

TABLE 12-13. EFFECTS OF CHRONIC EXPOSURE TO SO_x AND SOME PM

Concentration	Duration	Species	Results	Reference
Various combinations of SO_2, H_2SO_4 (0.5 to 3.4 µm, MMD), and fly ash (3.5 to 5.9 µm, MMD): SO_2, H_2SO_4 + fly ash, SO_2 + H_2SO_4, SO_2 + H_2SO_4 + fly ash	18 mo, continuous	Monkey	No significant effects on hematology or pulmonary function tests during exposure. At end of exposure to SO_2 + H_2SO_4 lungs had morphological alterations. Exposure to SO_2 + H_2SO_4 + fly ash had similar alterations; thus fly ash did not enhance effect. Exposure to H_2SO_4 + fly ash had slight alterations. (See pp 12-56 and 58 for details on size and concentration.)	Alarie et al., 1975
0.9 mg/m³ H_2SO_4 (0.49 µm, MMD); 0.08 mg/m³ H_2SO_4 (0.54 or 2.23 µm, MMD) + 0.45 mg/m³ fly ash (3.5 or 5.31 µm, MMD)	12 mo, continuous	Guinea pig	No significant effects on hematology, pulmonary function, or morphology	Alarie et al., 1975
Approximately 0.56 mg/m³ fly ash in combination with SO_2 at 0.28, 2.62, or 13.1 mg/m³ (0.11, 1, or 5 ppm).	78 wk, continuous	Monkey	No effects on pulmonary function, hematology, or morphology	Alarie et al., 1973b
Approximately 0.56 mg/m³ fly ash in combination with SO_2 at 0.28, 2.62, or 13.1 mg/m³ (0.11, 1, or 5 ppm)	52 wk, continuous	Guinea pig	No effects on pulmonary function, hematology, or morphology	Alarie et al., 1973b
13.4 mg/m³ (5.1 ppm) SO_2, or 0.89 mg/m³ H_2SO_4 (90% <0.5 µm in diameter), or to a combination of the two	21 h/day, 620 days	Dog	After 225 days, dogs receiving H_2SO_4 had a lower diffusing capacity for CO than those that did not receive H_2SO_4. In the SO_2-exposed group, pulmonary compliance was reduced and pulmonary resistance was increased compared to dogs that did not receive SO_2. Dogs not pre-exposed to NO_2 who received SO_2 + H_2SO_4 had a smaller residual volume than all other dogs. After 620 days, SO_2 increased mean nitrogen washout, but no hematological or morphological changes occurred. H_2SO_4 decreased: diffusing capacity for CO, residual volume, net lung volume, total lung capacity, inspiratory capacity functional residual capacity, total lung weight and heart rate. Total expiratory resistance increased.	Lewis et al., 1969, 1973

TABLE 12-13 (continued)

Concentration	Duration	Species	Results	Reference
(see Table 12-12)	16 h/day, 68 mo	Dog	Residual volumes increased in group receiving R + SO_x compared to CA, which showed increase compared to SO_x group. The I + SO_x group had higher total expiratory resistance. Thirty-two to 36 mo after exposure ceased, the SO_x group had increases in lung weight, total lung capacity, and displaced volume of the processed right lung and loss of cilia in the conducting airways. SO_x + R had loss of cilia and squamous metaplasia. Exposure to R + SO_x and I + SO_x produced nonciliated bronchiolar cell hyperplasia, increased interalveolar pores, and alveolar air space enlargement.	Lewis et al., 1969, 1973
5.24 mg/m³ (2 ppm) SO_2, or 0.56 mg/m³ carbon (1.8 to 2.2 µm, MMD), or in combination	100 h/wk, 192 days	Mouse	For the pulmonary immune system, only exposure to the combination caused significant effects; the systemic immune system was affected in all 3 exposure groups. Carbon and carbon + SO_2 were more effective than SO_2, although SO_2 did cause significant effects.	Zarkower, 1972
1.4 mg/m³ H_2SO_4 plus 1.5 mg/m³ carbon (0.4 µm, mean particle diameter), or 1.5 mg/m³ carbon only (0.3 µm, mean particle diameter)	3 h/day, 5 day/wk, 20 wk	Mouse	Altered the immune system. Morphological changes observed; more severe with carbon only exposure.	Fenters et al., 1979
1.1 mg/m³ H_2SO_4 (0.12 µm, mean size), or 1.5 mg/m³ carbon (0.3 µm, mean size), or in combination	3 h	Hamster	Ciliary beat frequency depressed after H_2SO_4, but not carbon, exposure. The combination produced similar effects, but recovery occurred by 48 h post-exposure. H_2SO_4 + carbon resulted in more tissue destruction than either pollutant alone.	Schiff et al., 1979

1 ppm SO_2 = 2.62 mg/m³.

requiring concentrations in excess of 100 mg/m^3 to produce changes in glycoprotein secretion. The DNA, RNA, and protein content of the lung increased in the group exposed to O_3 and H_2SO_4 aerosols, while the O_3-exposed group had only a small increase and the H_2SO_4 group had none.

Grose et al. (1980) investigated the interaction of H_2SO_4 and O_3 on ciliary beat frequency in the trachea of hamsters. A 2-h exposure to 0.88 mg/m^3 H_2SO_4 (0.23 µm, VMD) significantly depressed ciliary beat frequency. By 72 h after exposure, recovery had occurred. Hamsters exposed to 0.196 mg/m^3 (0.1 ppm) O_3 for 3 h were not significantly affected; however, when animals were exposed in sequence, first to O_3 and then to H_2SO_4, ciliary beat frequency was decreased significantly, but to a lesser extent than that caused by H_2SO_4 alone. Analysis showed that antagonism ($p < 0.05$) occurred in this sequential exposure.

Gardner et al. (1977a) found that the sequence of exposure to H_2SO_4 aerosols and O_3 altered the response of mice to airborne infections. Mice were exposed alone or in sequence to 0.196 mg/m^3 (0.1 ppm) O_3 for 3 h and to 0.9 mg/m^3 H_2SO_4 aerosol (0.23 µm, VMD, $\sigma_g = 2.4$) for 2 hours. When given alone, neither pollutant caused a statistically significant increase in the mortality to a subsequent infection with S. pyogenes. When the pollutants were given sequentially, a significant increase in mortality occurred only when O_3 was given immediately before exposure to H_2SO_4, and the response was additive. The reverse procedure had no effect on mortality due to S. pyogenes infections. Because photochemical oxidants and SO_x often coexist in polluted air, these studies are of practical importance. The question of the temporal sequence has been poorly investigated. Simple mechanisms to predict this additive response sequence are not apparent. The results are opposite those of the Grose et al. (1980) study described above with the tracheal model that showed that sequential exposure to O_3 and H_2SO_4 had an antagonistic effect. The reasons for this difference are not known. The infectivity model, however, is thought to reflect alveolar level effects (Gardner and Graham, 1977), whereas the ciliary beat frequency model is a measure of effects at the level of the trachea. In addition, different animal species were used. These findings also indicate the complexity of interaction effects and the need to exercise care in extrapolating the effects of pollutants from one parameter to another and one species to another (see Table 12-14).

12.5 CARCINOGENESIS AND MUTAGENESIS OF SULFUR COMPOUNDS AND ATMOSPHERIC PARTICLES

Attempts have been made for several decades to correlate various indices of particulate air pollution with the development of cancer in man. In some cases, a positive association has been found between increased community air pollution and cancer of the lungs and/or gastrointestinal tract (see Chapter 14). This has led to suspicions concerning the chemical nature of that portion or portions of airborne PM that may be contributing to an excess of human cancer. As a result, at least three classes of potential etiologic agents have been studied in animals: organic matter (including polycyclic hydrocarbons) that is adsorbed to suspended particles; sulfur oxides; and trace metals.

TABLE 12-14. EFFECTS OF INTERACTION OF SO_x AND O_3

Concentration	Duration	Species	Results	Reference
10 mg/m^3 (1 µm, MMD) H_2SO_4 aerosol, or 3.9 mg/m^3 (2 ppm) O_3, or combination of the two	6 h/day, 2 or 7 days	Rat and Guinea pig	No synergism in effect on ratio of lung to body weight. Histological lesions were those ascribed to O_3 alone.	Cavender et al., 1977
10 mg/m^3 0.83 µm MMAD, σ_g = 1.66) H_2SO_4 aerosol, or 1.02 mg/m^3 (0.52 ppm) O_3, or combination of the two	6 h/day, 5 day/wk, 6 mo	Rat and Guinea pig	Morphological alterations due to O_3 alone	Cavender et al., 1978
1 mg/m^3 H_2SO_4 aerosol and 0.78 to 0.98 mg/m^3 (0.4 to 0.5 ppm) O_3	3 days, continuous	Rat	Synergistic effects. Glycoprotein synthesis was stimulated in tracheal ring explants; lung DNA, RNA, and protein content increased.	Last and Cross, 1978
0.196 mg/m^3 (0.1 ppm) O_3; 0.9 mg/m^3 H_2SO_4 aerosol (0.23 µm, VMD, σ_g = 2.4) exposed alone or in sequence	3 h, O_3; 2 h, H_2SO_4	Mouse	Significant increase in mortality in response to airborne infections only when O_3 was given immediately before exposure to H_2SO_4; the response was additive.	Gardner et al., 1977a
0.196 mg/m^3 (0.1 ppm) O_3; 0.88 mg/m^3 H_2SO_4 aerosol (0.23 µm, VMD) exposed alone or in sequence.	3 h, O_3; 2 h, H_2SO_4	Hamster	H_2SO_4 depressed ciliary beat frequency; Recovery had occurred by 72 h after exposure. Sequential O_3 then H_2SO_4 exposure decreased ciliary beat frequency significantly but to a lesser extent than that caused by H_2SO_4 alone. O_3 exposure had no effect.	Grose et al., 1980

Test systems for the bioassay of potential mutagens and carcinogens are diverse, ranging from the measurement of chemically-induced reverse mutations in bacteria to the frank production of carcinomas by administration to mammals. It is commonly believed, however, that fundamental similarities exist between the molecular mechanisms of both mutagenesis and carcinogenesis. This assumption is based on the theory that chemical interaction with DNA and/or other critical cellular macromolecules initiates a mutagenic or carcinogenic transformation.

Because of the assumed relationship between molecular events involved in mutagenesis and carcinogenesis (Miller, 1978), the demonstration of mutagenic activity for a substance is generally taken as strong presumptive evidence for the existence of carcinogenic activity. Therefore, it is believed that an investigation of the mutagenicity of a substance may be predictive of its carcinogenic potential, and may serve as an early warning of a possible threat to human health in cases where positive results are obtained.

12.5.1 Airborne Particulate Matter

12.5.1.1 In Vitro Mutagenesis Assays of Particulate Matter--Organic material associated with airborne particles has been investigated to some extent for its mutagenic potential. In these studies, PM is limited experimentally to that which is retained by the filter medium used, e.g., glass fiber, paper...etc. (Clark et al., 1980; Lee et al., 1980; Pitts et al., 1978). The particles that have created most interest are those with a carbonaceous core. These particles, because of their large surface area, adsorb many organic compounds, some of which are known to be mutagenic and carcinogenic, e.g., benzo(a)pyrene (B(a)P). Because of the small size (0.2-0.3 μm mean diameter) of many of the particles, they can be deposited in the pulmonary regions of the lung (Verrant and Kittelson, 1977; Wolff et al., 1981b) where the adsorbed organic material can desorb into the alveolar fluid and enter the associated tissue. The ability of serum proteins to leach mutagens off particles has been demonstrated using horse serum and coal fly ash (Crisp et al., 1978; Brooks et al., 1979; King et al., 1981; Clark and Virgil, 1980; Wang and Wei, 1980).

A number of studies using the Ames *Salmonella* mutagenicity assay and other *in vitro* tests have been conducted with fractionated extracts of PM from urban air to obtain information on the chemical nature of the mutagens present (Dehnen et al., 1977; Teranishi et al., 1978; Moller and Alefheim, 1980; Tokiwa et al., 1980; Huisingh, 1981; Kolber et al., 1981; Ohnishi et al., 1980). Estimates have been made as to the relative mutagenicity of each extract; however, due to the possible interaction among the many compounds present in any fraction of the extracts, the only conclusion that can be drawn is that both the moderately polar and neutral fractions contain significant portions of the total mutagenic activity. The former is the more potent direct-acting mutagen, while chemical derivatives of polycyclic aromatic hydrocarbon (PAH) compounds are contained in the neutral fraction (Daisey, 1980; Daisey et al., 1980; Kotin et al., 1955; Goff et al., 1980; Falk and Steiner, 1952; Henderson et al., 1981). At present, the identity of the compounds that act as direct mutagens is uncertain.

In a similar manner, various fractions were also extracted from particles emitted from a coal powered electric plant (Crisp et al., 1978; Kubitschek et al., 1979; Clark and Hobbs, 1980; Fisher et al., 1979), gasoline engines (Wang et al., 1978), and light- and heavy-duty diesel engines (Huisingh et al., 1977). The extracts obtained from these sources were direct-acting frame-shift mutagens. Both the heavy- and light-duty diesel engine study was fractionation carried out on the crude extract (Wang et al., 1978; Dukovich et al., 1981; Li et al., 1980). A review of diesel engine PM is available (Santodonato, 1978).

The Salmonella assay has also been used in an attempt to define air quality by measuring the mutagenic potential of total airborne particulates. Tokiwa et al. (1977) compared the number of revertants per μg of PM collected in the industrial area of Ohmata with that collected in the residential area of Fukuoka, Japan. In a similar manner, Pitts et al. (1977, 1978) compared eight urban samples in the California South Coast Basin with one collected in a rural area of the San Bernadino mountains. In both cases, the mutagenic activity was less in the residential and rural areas compared to the urban areas. Mutagenic potential was also determined in a quantitative manner for a variety of air samples collected in Chicago (Commoner et al., 1978).

Caution must be exercised when comparing in a quantitative manner results of Ames assays on complex environmental mixtures. Indirect mutagenesis is extremely difficult to quantitate, since microsomal oxidation to nonreactive as well as reactive compounds occurs (Kaden et al., 1979; Skopek et al., 1979). Mixtures of direct and indirect mutagens may not produce an additive result. For any valid comparison there has to be nearly complete separation of these two types of mutagens (Commoner et al., 1978). Also, the effects on mutagenesis of synergism and antagonism among compounds in complex mixtures has not been investigated adequately. In the case of complex mixtures obtained from tar sand, for example, the mutagenic activity of the known mutagen, 2-aminoanthracene, was greatly inhibited by interaction with the mixture (Shahin and Fournier, 1978). For these reasons, a quantitative assessment of air quality is not readily obtainable with the use of the Ames Salmonella mutagenicity assay.

The data obtained with mammalian cell transformation assays (Rudd, 1979; Li, 1981; Liber et al., 1979) generally support the conclusions derived from the Ames Salmonella assays (Heidelberger, 1978; Sivak, 1979). There appear to be a variety of biologically active agents present in the extracts of airborne PM, and these agents are of both a polar and nonpolar nature. The identity of these compounds is unknown; however, the activity present is greater than that which could be accounted for only by the PAH present in the samples. Since it is presently unclear how the process of transformation in virus-infected cells relates to the process of chemical carcinogenesis, cell transformation assays should be considered in the same way as Ames assays, i.e., as only an indicator of the presence of biologically active compounds.

The dominant lethal assay of Epstein et al. (1972) is the only short-term in vivo assay performed on airborne particulate extracts. The water-soluble and benzene-soluble fractions produced no fetal deaths or preimplantation losses beyond control limits. On the other hand, the oxygenated fraction showed significant fetal deaths and decreased total implants.

12.5.1.2. Tumorigenesis of Particulate Extracts--As early as the 1930s it was realized that increasing amounts of air pollution may correlate with the increasing rate of human lung cancer. Some of the earliest in vivo experiments dealt with the repeated exposure of mice to clouds of soot, followed by autopsy examination for tumors at the end of their natural lifespan. A number of different kinds of soot have been chosen for these studies due to their significant contribution to airborne PM. Upon bioassay of soot from chimneys (Campbell, 1939; Seelig and Benignus, 1938), motor exhaust (Campbell, 1939), and airborne PM collected in the vicinity of a factory and roadway (McDonald and Woodhouse, 1942), a slight increase over control levels in the number of lung tumors was observed. Only in the case where road dust from a freshly tarred road was used were there significant increases, with 57 percent of the experimental and 8 percent of the control group having lung tumors (Campbell, 1934). However, 5 yrs later dust from the same road, which had not been retarred, was again tested and only 8 percent of the experimental group and 1.4 percent of the control group developed lung tumors (Campbell, 1942). Although these studies have all attempted to demonstrate the potential of airborne PM to cause lung tumors, the results obtained are ambiguous due to the low tumor incidence and the small size of the animal groups.

Among the various compounds associated with airborne particles, PAH (polycyclic aromatic hydrocarbons) have received the most attention as potentially carcinogenic. PAHs were the first compounds shown to be associated with carcinogenesis. Carcinogenic PAH are still distinguished by several unique features:

1. several compounds of this class are among the most potent animal carcinogens known to exist, producing tumors by single exposures to microgram quantities.
2. they act both at the site of application and at organs distant from the site of absorption.
3. their effects have been demonstrated in nearly every tissue and species tested, regardless of the route of administration.

The most widely studied PAH, benzo(a)pyrene (B(a)P), is ubiquitous in the environment and produces tumors in animals that closely resemble human carcinomas.

The production of lung tumors in animals has been difficult. PAHs such as B(a)P are mildly carcinogenic in the respiratory tract using intratracheal instillations. Certain inorganic materials (iron ore, carbon, asbestos) have been found to potentiate this effect of B(a)P in hamsters (Pylev and Shabad, 1973; Stenback et al., 1973, 1976; Saffioti et al., 1968) and produce bronchogenic carcinomas with a morphology similar to human lung cancer. Organic extracts of airborne particulates readily cause tumors when injected subcutaneously. Sarcomas have been produced in mice using the benzene extracts of PM collected from an urban area (Leiter and Shear, 1942; Leiter et al. 1942). The tumor incidence was low in these initial studies, with only 8 percent of the mice developing tumors by the end of the study; however, none of the control mice had sarcomas. In one later study, when particles were collected in the vicinity of a petrochemical plant, the tumor incidence was as high as 61 percent (Rigdon and Neal, 1971). Even in this case of high tumor production, no increase in the incidence of tumors over the spontaneous rate was observed in any organ of the animal distant to the site of injection. Only when neonatal mice were injected subcutaneously with particulate extracts did tumors appear distant from the injection site (Epstein et al., 1966), with a very high incidence of hepatomas (83 percent) and multiple pulmonary adenomas (67 percent). Remote tumor formation after subcutaneous injection of neonatal mice was confirmed with both the crude extract of particles collected in New York City and subfractions of this extract; the predominant tumors were again hepatomas (Asahina et al., 1972).

The carcinogenic nature of extracts of PM has also been demonstrated by painting the skin on the backs of mice. With repeated application (three times per wk for the life of the animal) of the benzene extract of particulates collected in the Los Angeles area, papillomas were formed that subsequently progressed to carcinomas (Kotin et al., 1954). Papillomas first appeared after 465 days, and at the time the data were presented, 42 percent of the mice had developed tumors. Although papillomas and carcinomas of the skin were the most commonly observed tumors, lung tumors have also been noted after skin application (Clemo et al., 1955). Among the different methods of administering particulate extracts to the mouse for bioassay, skin painting yields the highest tumor incidence, with greater than 90 percent of the surviving animals in some cases developing tumors.

In subsequent studies, the phenomenon of two-stage tumorigenesis was used to characterize further the biological activity in airborne PM. In two-stage tumorigenesis, an initiator is an agent (usually a carcinogen) that, when applied in a single dose to the skin of a mouse, does not produce tumors at the applied concentration, but predisposes the skin so that later repeated application of a promoter (an agent that by itself will not produce tumors) will cause the formation of tumors. A complete carcinogen is one that, if applied in sufficient concentration, can produce tumors by itself. Extracts of airborne particles from Detroit were fractionated, and the fractions examined for complete carcinogenicity and tumor initiating and promoting activity (Stern, 1968; Wynder and Hoffman, 1962). When applied to the skin of Swiss ICR female mice, only the whole extract and the aromatic fraction proved to be a complete carcinogen, while the insoluble, acidic, aliphatic and oxygenated fractions produced no tumors (there was insufficient basic fraction to perform the assay).

To examine the aromatic fraction for initiating activity, it was applied to the backs of mice in a subtumorigenic dose followed by repeated application of the known promoter: croton oil. Tumor initiating activity corresponded in a general way to the B(a)P content of the fraction. (The other fractions of the particulate extract were not tested for initiating activity.) It should be noted that an initiator does not necessarily have to be a complete carcinogen, although most if not all complete carcinogens will initiate if applied at a low dose where their complete carcinogenic action is not apparent. For this reason, it is possible that some of these fractions could have initiated activity even though they did not act as complete carcinogens when first tested. In addition, both the acidic and neutral fractions exhibited tumor promoting activity in the presence of the known initiator: 7, 12-dimethylbenz(a)anthracene (DMBA). The relevance of two-stage skin carcinogenesis to environmentally-caused cancer in other organs, however, is not known.

Several contributing sources of airborne PM (e.g., gasoline and diesel engines and the soot from coal and oil burning furnaces) have been examined individually and shown to produce tumors. Extracts of PM from gasoline engines show carcinogenic activity when painted on the backs of mice (Brune, 1977; Wynder and Hoffman, 1962) and injected subcutaneously (Pott et al., 1977). Extracts from diesel engines have shown tumorigenic activity in some studies but not in others; the same holds true for extracts of chimney soot where activity was shown in some instances (Campbell, 1939) while not in others (Mittler and Nicholson, 1957). The discrepancies among these results could be due to qualitative and/or quantitative differences in the nature of the organic compounds adsorbed to the particles or difference in assay systems. Differences may have existed in the operating parameters of the generating source, or variations in PM collection procedures. For example, the mode of operation of diesel engines (the load under which the engine was run), the type of fuel and the temperature at which the particles were collected all affect the biological activity of the sample. With soot collected from chimneys, temperature is an important consideration. The organic material on PM is generated in the gaseous phase, while condensation on nuclei occurs at lower temperatures. Unless particles are collected under similar conditions, disparities will exist in their chemical composition and biological activity.

Taken together, it is apparent that all the major types of airborne PM may contain adsorbed compounds that are mutagenic and/or carcinogenic to animals and may contribute in some degree to the incidence of human cancer associated with exposure to urban air pollution.

12.5.2 Potential Mutagenic Effects of Bisulfite and Sulfur Dioxide

Bisulfite addition to cytosine can result in deamination to form uracil (Shapiro, 1977; Fishbein, 1976). The result would be conversion of guanine-cytosine in DNA to adenine-thymine sites. Transamination of cytosine can also occur through reaction of an amine with cytosine-bisulfite adduct. Since the nucleus is rich in polyamines, transamination would appear to be a likely event; however, these reactions of cytosine occur most readily in high (1 M) concentrations of bisulfite. Because of the nature of the reactivity of bisulfite with cytosine, the potential mutagenic properties of bisulfite and SO_2 have been examined. Such experiments have been reviewed by Shapiro (1977) and Fishbein (1976). Microbial experiments with high concentrations of bisulfite in acid solutions *in vitro* have produced mutations (see Table 12-15).

TABLE 12-15. POTENTIAL MUTAGENIC EFFECTS OF SO_2/BISULFITE

Concentration	Bisulfite	Organism	End Point	Response	Comments	Reference
	0.9 M HSO_3^- pH 5.0	Phage T4-R11 System	GC→AT or deamination of cysocine	+	-	Summers and Drake, 1971
	3 M HSO_3^- pH 5-6	Phage T4-R11 System	deamination of cytosine	±	Poor dose response	Hayatsu and Miura, 1970, Iida et al., 1974
	1 M HSO_3^- pH 5.2	E. coli K12 & K15	GC→AT or deamination of cytosine	+	-	Mukai et al., 1970
	5 x 10^{-3} M HSO_3^- pH 3.6	S. cerevisiae	Point Mutation	+	-	Dorange and Dupuy, 1972
	0.04 or 0.08 M	D. melanogaster	Point Mutation	-	May not be bioavailable	Valencia et al., 1973
1310 mg/m³ SO_2 (500 ppm)		Hela cells (Human)	Cytotoxicity	+	-	Thompson and Pace, 1962
13.1 - 105 mg/m³ SO_2 (5 - 40 ppm x 3 min)		Mouse fibroblasts & Peritoneal macrophages	-		-	Nulsen et al., 1974
14.9 mg/m³ SO_2 (5.7 ppm)		Human lymphocytes	Point Mutation	-	-	Kikigawa and Iizuka, 1972
			Chromosomal aberrations	-	-	
			Cytotoxicity	+	-	

1 ppm SO_2 = 2.62 mg/m³.

These conditions would be similar to those favoring deamination of cytosine. On the other hand, experiments conducted at low concentrations (> 10^{-3}M bisulfite) and neutral pH (7-7.4) have not provided clear-cut evidence of mutagenesis. The microbial assays were not done with strains of *Salmonella* known to be sensitive to mutagens (Ames assays). Background mutation rates, mechanisms of error-prone repair, and corrections for cytotoxicity were not studied. Negative experiments have been reported when insects (Drosophila) (Valencia et al., 1973) and mammals (mice) were exposed. Cytotoxicity, rather than mutagenicity, appears when cultured animal and human cells (Thompson and Pace, 1962; Nulsen et al., 1974; Kikigawa and Iizuka, 1972; Schneider and Calkins, 1971; Timson, 1973) are exposed to sulfite. (See Table 12-15 for summary.)

12.5.3 Tumorigenesis in Animals Exposed to Sulfur Dioxide or Sulfur Dioxide and Benzo(a)pyrene

Tumorigenesis after exposure to SO_2 alone or to SO_2 and an aerosol of B(a)P has been examined. For example, Peacock and Spence (1967) examined the lungs and other organs of mice exposed over their lifetimes (300 days) in a 180-liter chamber into which 500 ppm SO_2 was introduced at a rate of 20 ml/min for 5 min, 5 days/week. They found an increase in primary pulmonary neoplasia in the males (n=35) from 31 percent in the control group to 54 percent in the SO_2-exposed group, and in the females (n=30) from 17 to 43 percent. Furthermore, while SO_2 did not affect the incidence of malignant tumors in males, the incidence of primary lung carcinoma in females increased from 0 to 18 percent in the exposed group.

Unfortunately, the concentration of SO_2 used cannot be determined accurately from the paper and thus, no concentration-related effects can be deduced. In addition, the statistical analyses reported are vague and the significance of the observed increases in lung carcinoma is, therefore, questionable. In view of these shortcomings, EPA undertook a reanalysis of the data reported in Peacock and Spence (1967). This reanalysis, which is described in the memo attached as an appendix to this chapter, used a one-sided Fisher's exact test to determine that SO_2 did indeed increase the incidence of primary lung carcinoma in females (p=0.056), as well as the incidence of primary lung adenomas in males (p=0.065) and females (p=0.011). Despite these findings, it would still appear that Peacock and Spence's conclusion that this particular study does not "justify the classification of SO_2 as a chemical carcinogen as generally understood" remains valid.

Lung tumors or other significant pathological effects were not observed in hamsters exposed for 98 wk to 26.2 mg/m^3 (10 ppm) SO_2 plus 10 mg/m^3 B(a)P for 6 h/day, 5 days/wk for 534 exposure days, or to 9.17 mg/m^3 (3.5 ppm) SO_2 plus 10 mg/m^3 B(a)P for 1 h/day, 5 days/wk for 494 exposure days, or to a combination of the two regimens (Laskin et al., 1970). When rats were exposed in the same fashion (Laskin et al., 1970), however, lung squamous cell carcinoma was found in 23.8 percent of the animals exposed to the combination of the two regimens described above and in 9.5 percent of animals exposed to 10 mg/m^3 B(a)P plus SO_2 for 1 h/day. Renal metastasis also occurred.

This study was subsequently extended to lifetime (exact time not specified) exposures (5 days/wk) of rats (Laskin et al., 1976). Exposure to air alone (n=15) or to 26.2 mg/m^3 (10 ppm) SO$_2$ (n=15) for 6 h/day caused no squamous cell carcinoma. A 1-h/day exposure to 10 mg/m^3 B(a)P caused cancer in 1 of 30 (3.3 percent) rats. A 6-h/day exposure to 26.2 mg/m^3 (10 ppm) SO$_2$ plus a 1-h/day exposure to 10 mg/m^3 B(a)P resulted in a cancer incidence of 6.7 percent (2 of 30). When animals received a combination of 10 mg/m^3 B(a)P and 10.5 mg/m^3 (4 ppm) SO$_2$, 4 of 45 (8.9 percent) of the rats had cancer. The highest incidence (9 of 46, 19.6 percent) was found in animals exposed for 6 h/day to 26.2 mg/m^3 (10 ppm) SO$_2$ plus a combination of 10 mg/m^3 B(a)P and 10.48 mg/m^3 (4 ppm) SO$_2$ for 1 h/day.

The biological significance of these studies (Laskin et al., 1970, 1976) is complex and difficult to interpret, particularly since statistical analyses were not reported. In an attempt to clarify the matter, EPA undertook a reanalysis of the latter study of Laskin et al. (1976). The EPA statistical analysis is described in the memo attached as an appendix to this chapter. Using a multiple probit approach, it was determined that, while the cancer incidence increases in response to exposure to SO$_2$ alone or B(a)P alone were not statistically significant ($p=0.116$ and $p=0.113$, respectively), the increase due to the combination of the two was indeed significant ($p=0.005$). However, given the lack of experimental details, especially with the experimental design used, it is very difficult to come to a definitive conclusion about the carcinogenicity of SO$_2$ and B(a)P as administered in this protocol.

12.5.4 Effects of Trace Metals Found in Atmospheric Particles

Among the numerous trace metals found in the atmosphere, systemic toxicity has been demonstrated following inhalation of lead (Office of Research and Development, 1977), mercury (Hammond and Beliles, 1980), arsenic (NAS, 1977), asbestos (NAS, 1971), and cadmium (Hammond and Beliles, 1980). In addition, certain compounds of some of these (e.g., arsenic, beryllium, cobalt, and nickel) have been found to produce tumorigenic effects under specific, nonrespiratory laboratory exposure conditions (Furst and Hard, 1969; IARC, 1972, 1973, 1976; Lau et al., 1972; Stoner, et al., 1976; Sunderman, 1978, 1979). Limited evidence also points to compounds of molybdenum and manganese as possible tumorigens (Clemo and Miller, 1960).

Although trace metals are ubiquitous in the environment, their levels are generally so low that it is difficult to predict the magnitude of carcinogenic risk in community settings. This problem is compounded by the fact that clear dose-response relationships have not been well defined for most carcinogenic metals. At present, therefore, it is not possible to predict confidently the quantitative relationship of trace metals to the production of cancer due to low-level exposures to particulate air pollution.

12.6 CONCLUSIONS

12.6.1 Sulfur Dioxide

Once inhaled, SO$_2$ appears to be converted to its hydrated forms, sulfurous acid, bisulfite, and sulfite. The rate of absorption and removal of inhaled SO$_2$ varies with species, but it is at least 80 percent of the inhaled amount at relatively high concentrations.

The metabolism of SO_2 is predominantly to sulfate and is mediated by the enzyme sulfite oxidase. Since sulfite oxidase is a molybdenum-containing enzyme, dietary factors could influence the function of the enzyme in man. No conclusive evidence has yet been reported. The reaction of bisulfite with serum proteins to form S-sulfonates is rapid. The S-sulfonates are remarkably long-lived (t 1/2 = 4.1 days in rabbits), supplying a circulating pool of bisulfite that can reach all tissues. Since some circulating S-sulfonates decompose to SO_2 that is exhaled, S-sulfonates can donate their bisulfite content to distal tissues.

An immediate effect of acute (≦ 1 h) SO_2 inhalation is either a decrease in respiratory rate or an increase in resistance to flow within the lung. The decrease in respiratory rate depends on afferent conduction through the Vth or IXth cranial nerve following activation of receptors in the nose and upper airways. Nasal air flow is decreased. The response is transient in nature and occurs at 44.5 mg/m^3 (17 ppm) SO_2. Lower concentrations were not tested.

The increased resistance to flow on inhalation of SO_2 is mediated through receptors in the bronchial tree and persists during continued exposure. With this physiological parameter, lower concentrations of SO_2 have been observed to cause reproducible changes in respiration. The increased resistance to flow of air in the lung following SO_2 inhalation represents the activation of an autonomic reflex arc through the vagus nerves. The same reflex arc occurs in man. The reflex is cholinergic since atropine blocks the reflex, presumably at preganglionic synapses. The guinea pig is the most sensitive animal in which to measure airway resistance, with significant changes in pulmonary resistance to airflow occurring on the inhalation of concentrations as low as 0.42 mg/m^3 (0.16 ppm) SO_2 for 1 hour. Chronic exposures have produced alterations in pulmonary function in cynomolgus monkeys, but only at concentrations greater than 13.1 mg/m^3 (5 ppm). Dogs exposed to 13.4 mg/m^3 (5.1 ppm) SO_2 for 21 h/day for 225 days had increased pulmonary flow resistance and decreased compliance. Lower concentrations were not examined. It should be remembered that SO_2 appears to cause its immediate bronchoconstrictive effect through action on airway smooth muscles, as evidenced by the antagonism of the SO_2-initiated bronchoconstriction by isoproterenol in man and animals. Since smooth muscles adapt or fatigue during long-term stimulation, chronic exposure to SO_2 is not likely to evidence bronchoconstriction equivalent to that occurring on short-term exposure. Alterations in pulmonary function after chronic exposure to SO_2 are likely to occur through other mechanisms, such as morphological changes in the airways or hypersecretion of mucus, which will result in narrowing the airway. Concentration, rather than duration of exposure, seems to be the most important parameter in determining responses to SO_2, whether the response is measured as a histopathological lesion or as a permanent alteration in respiration. There is no theoretical hypothesis available at present to integrate the short-term effects observed with 1-h exposures and the effects of long-term exposures of several months.

In rats, histopathological effects of SO_2 alone are confined to the bronchial epithelium, with most of the effects occurring on the mucus secreting goblet cells. Goblet cell hypertrophy occurs on chronic exposure of rats, leading to the suggestion that SO_2 produces a chronic bronchitis similar in many respects to that in man. Repeated exposure to a critical concentration of SO_2 (not less than 131 mg/m^3 or 50 ppm) may be needed to produce the chronic bronchitis. While SO_2-produced chronic bronchitis in rats is similar to that in man and is a useful model for the study of bronchitis, no evidence exists that chronic bronchitis is produced in man from ambient concentrations of SO_2.

The nasal mucosa of mice (particularly those with upper respiratory pathogens) was altered by a 72-h exposure to 26.2 mg/m^3 (10 ppm) SO_2. Continous exposure to 0.37 to 3.35 mg/m^3 (0.14 to 1.28 ppm) SO_2 for 78 wk did not cause any significant morphological alterations in the lungs of monkeys. The effects of near-ambient concentrations of SO_2 on the morphology and function of the nasal mucosa are not known.

Some pulmonary host defense mechanisms are also affected by SO_2 exposure. After 10 and 23 days of exposure (7 h/day, 5 days/wk) to 0.26 mg/m^3 (0.1 ppm)), clearance of particles from the lower respiratory tract was accelerated in rats. At a higher concentration of SO_2 (2.62 mg/m^3, 1 ppm) there was an initial acceleration (at 10 days), followed by a slowing at 25 days. A 5-day (1.5 h/day) exposure to 2.62 mg/m^3 (1 ppm) reduced tracheal mucus flow in dogs, but a longer exposure to this concentration caused no changes in ciliary beat frequency of rats. These aberrations in tracheal clearance and mucus flow in several species are consistent with the profound effects of higher concentrations of SO_2 on mucus glands in rats.

Antiviral defenses were altered by a 7-day continous exposure to 18.3 to 26.2 mg/m^3 (7 to 10 ppm) SO_2, as evidenced by an increase in viral pneumonia. The combined exposure to SO_2 and virus produced weight loss on exposure to concentrations as low as 9.43 mg/m^3 (3.6 ppm) SO_2. Mice exposed to 13.1 mg/m^3 (5 ppm) SO_2 for 3 h/day for 1 to 15 days or for 24 h/day for 1 to 3 mo did not have increased susceptibility to bacterial lung disease. A variety of changes in the humoral immune response of mice exposed for up to 196 days to 5.24 mg/m^3 (2 ppm) have been reported.

Sulfur dioxide and bisulfite have been reported to be mutagenic in microbial test systems (*E. coli* and yeast systems) at nonphysiological (acid) pH, but the relevancy of inhaled SO_2 as a mutagen is not clear. A mechanism for the mutagenicity of SO_2 could be the deamination of cytosine at high concentrations. Free radical reactions breaking glycosidic bonds in DNA may be responsible at low concentrations. The potency of bisulfite in these *in vitro* systems is moderate to weak when compared to agents such as nitrosamines or polycyclic aromatic compounds. To date, experiments testing for mutagenicity of bisulfite in mammals have been equivocal. On the basis of present evidence, one can not decide whether or not bisulfite, and hence SO_2, is a mutagen in mammals.

The potential carcinogenicity of SO_2 alone and in combination with B(a)P has also been examined. Unfortunately, the two key studies have not been replicated and both lacked experimental details and proper statistical analyses. While subsequent reanalyses of the data by

EPA revealed statistically significant increases in cancer, it is very difficult at present to come to a definitive conclusion about these studies. However, SO_2 must remain suspect as a carcinogen or cocarcinogen in view of these reanalyses and the positive results of mutagenicity assays.

12.6.2 Particulate Matter

The chemical and physical diversity of the PM in the atmosphere presents a severe limitation on the scope of the conclusions presented here. Most of the evidence for adverse health effects of inhaled particles presented in this chapter relates to compounds arising from SO_x (e.g., H_2SO_4, ammonium sulfate, metal sulfates, and related compounds). A brief treatment is presented for heavy metals and their compounds. A summary of data related to organic material associated with particles is also presented, focusing primarily on polycyclic organic compounds. Due to limitations of space, details dealt with in other criteria documents or recent major reviews on specific elements and the broader aspects of polycyclic organic compounds are presented by reference only. This chapter, then, should not be taken as a summary of all of the data available on the health effects of atmospheric particles, but rather as a selected summary related mostly to SO_x. The reader should refer to the more detailed reports before attempting to integrate the present limited material into the generic problem of the effects of atmospheric particles. Similarly, the subsequent section on the interactions between SO_x and PM relates to a limited scope of the atmospheric particles.

All inhalation studies of particles available for review in this document were conducted with sizes that could be expected to fall within the approximate size range of the alveolar envelope of deposition (<8 µm MMAD, depending upon the species exposed). A few in vitro or intratracheal instillation studies have been performed that compared the effects of a wide range of particles including those that occur in the atmospheric coarse mode particle sizes (>2.5 µm D_{ae}).

Reports disagree as to the potency of acute exposure to sulfate aerosols. Some investigators contend that H_2SO_4 is highly irritating, producing increases in pulmonary flow resistance at low concentrations. The increased resistance to airflow in the lung was directly proportional to the sulfate aerosol concentration inhaled. The bronchoconstriction produced by zinc ammonium sulfate was similar in many properties to that produced by histamine aerosols. Unlike SO_2-initiated bronchoconstriction, intravenous atropine had no effect. Inhaled or intravenous isoproterenol, however, blocked the zinc ammonium sulfate-induced bronchoconstriction. These data suggest that the zinc ammonium sulfate aerosol receptor and presumably other sulfate receptors are not identical to the SO_2 receptor. The two agents accordingly could act at separate sites in the lung. Histamine is implicated in the sulfate aerosol action more clearly than in the bronchoconstricting action of SO_2.

The lowest effective concentration of H_2SO_4 producing bronchoconstriction so far reported was 0.1 mg/m^3 (1 h) in the guinea pig. Particle size influenced the results in several ways, but the smaller sizes were generally more effective. Another study has observed an "all or none" response (increased airway resistance) in guinea pigs exposed for 1 h to 14.6, 24.3, or 48.3 mg/m^3 H_2SO_4. Exposure to lower concentrations (1.2 or 1.3 mg/m^3 H_2SO_4) caused no effects. Some of these conflicts may be due to differences in technique or in age or strain of guinea pig. Large interindividual differences in dose-response curves are characteristic for inhaled histamine. In man, dogs, cats, and guinea pigs, 100-fold differences in the bronchoconstrictive response to histamine have been observed. The dose-response curves for an individual subject are remarkably reproducible. In dogs and guinea pigs, the bronchoconstrictive response to histamine fell within a single log-normal distribution, despite the large interindividual differences in the dose of histamine required to elicit a specific response. These large interindividual differences could represent differences in "susceptibility" of different individuals, suggesting a small fraction of "susceptible" individuals; or they could represent a very flat dose-response curve for a single population. Currently, the data favor a single population hypothesis for histamine. The dose-response relationship for sulfate and H_2SO_4 aerosols is not adequate to distinguish between these two hypotheses. It is clear that large interindividual differences in response to inhaled aerosols are a characteristic of the biological response as measured by increased resistance to flow, regardless of the species used, and are not an artifact of the exposure or measurement system.

Age may also play in important part in this response, since young guinea pigs are more susceptible than older ones. For histamine sensitivity, age-dependence has been suggested as an analog of juvenile asthma, but human airway sensitivity does not seem to follow the same developmental pattern. Further research is needed to settle the question of special susceptibility of young animals and children.

For the effects of 1-h exposures of guinea pigs to SO_x from one laboratory, an apparent ranking of potency (for increased flow resistence) is as follows: H_2SO_4 > $ZnSO_4(NH_4)_2SO_4$ > $Fe_2(SO_4)_3$ > $ZnSO_4$ > $(NH_4)_2SO_4$ > NH_4HSO_4, $CuSO_4$ > $FeSO_4$, Na_2SO_4, $MnSO_4$. The latter three caused no effects.

The toxicology of H_2SO_4 is complicated by its partial concentration-dependent conversion to $(NH_4)_2SO_4$ and NH_4HSO_4 by ammonia in the breath or in the air of exposure chambers, due to excrement or exhalation. However, the actual concentrations of $(NH_4)_2SO_4$ and NH_4HSO_4 in the airways or chambers have not been measured definitively. Thus, comparing results of different H_2SO_4 studies is confounded, particularly since some neutralization would even be expected in the atmosphere of human exposure chambers (Kleinman et al., 1981). One theory for the irritating action of sulfates contends that sulfate salts can act to promote release of histamine or other mediators of bronchoconstriction and is supported by biochemical and pharmacological evidence in two species. Anionic release of histamine may play a role in the bronchial constriction as evidenced by the blockade with H-1 antihistamines. The effects of adrenergic agonists and antagonists suggest the involvement of tracheal smooth muscle. Certainly, the clearance of sulfurous acid, bisulfite, sulfite, and sulfate from the lung is influenced by the cations present in the aerosols inhaled simultaneously. Since polluted air is such a complex mixture of these aerosols, the question of the toxicity of ambient aerosols can not be approached on a simplistic basis by estimating toxicity from the acidity or sulfate content alone.

Chronic exposure to H_2SO_4 also produces changes in pulmonary function. Monkeys exposed to 0.48 mg/m^3 H_2SO_4 continously for 78 wk had altered distribution of ventilation early in the exposure period. Higher concentrations (2.43 and 4.79 mg/m^3 H_2SO_4) changed the distribution of ventilation and increased respiratory rate, but caused no effects on other pulmonary function measurements. A lower concentration (0.38 mg/m^3 H_2SO_4) caused no effects. Morphological changes occurred at the lowest concentration tested (0.38 mg/m^3 H_2SO_4). The effects appeared to be related to size of the particle as well as to concentration. Major findings at 2.43 mg/m^3 H_2SO_4 included bronchiolar epithelial hyperplasia and thickening of the respiratory bronchioles and alveolar walls. Guinea pigs exposed continously for 52 wk to 0.08 or 0.1 mg/m^3 H_2SO_4 had no effects on pulmonary function or morphology. Dogs that inhaled 0.89 mg/m^3 H_2SO_4 for 620 days (21 h/day) also had no morphological alterations; however, CO diffusing capacity, residual volume, and net lung volume were decreased. Several other changes were noted, including an increase in total expiratory resistance.

Sulfuric acid also alters mucociliary clearance, which is responsible for clearing the lung of viable or inanimate particles. These particles affect the ciliated airways during inhalation or reach this region as a result of alveolar clearance. A 1-h exposure of dogs to 0.5 mg/m^3 H_2SO_4 increased tracheal mucocilary transport, whereas 1 mg/m^3 H_2SO_4 depressed this rate. A 2- to 3-h exposure to 0.9 to 1 mg/m^3 H_2SO_4 also decreased tracheal ciliary beat frequency in hamsters. Lower concentrations (0.1 mg/m^3 H_2SO_4, 1 h/day, 5 days/wk) caused erratic bronchial mucociliary clearance rates in donkeys after several wk of exposure. Continued exposure of the donkeys that had not received preexposures caused a persistent slowing of bronchial clearance after about 3 mo of exposure. From these and other studies, it appears that repeated exposures to low concentrations of H_2SO_4 can slow mucociliary clearance. This might imply increased lung residence times of materials that would ordinarily be cleared.

Other host defense parameters (e.g., resistance to bacterial infection) are not altered by low concentrations of H_2SO_4, but are affected by metal sulfates. The apparent relative potency of various particles for increasing susceptibility to infectious (bacterial) respiratory disease has been determined in mice exposed for 3 h: $CdSO_4 > CuSO_4 > ZnNO_3$, $ZnSO_4 > Al_2(SO_4)_3 > Zn(NH_4)_2(SO_4)_2$. At concentrations > 2.5 mg/m^3, the following particles had no significant effects in this model system: H_2SO_4, $(NH_4)_2SO_4$, NH_4HSO_4, Na_2SO_4, $Fe_2(SO_4)_3$, $Fe(NH_4)_2SO_4$, $NaNO_3$, KNO_3, and NH_4NO_3.

The chemical composition of the sulfate aerosols determines their relative toxicities. For pulmonary irritants, the potency of a sulfate salt aerosol can be correlated with the permeability of the lung to that specific sulfate salt. However, it is evident that accurate estimates of the toxicity of complex aerosols occurring in urban air based solely on their sulfate contents are inappropriate, since the metallic ions often associated with them may also be toxic. Since urban air contains H_2SO_4, ammonium sulfate, and metallic sulfates in varying proportions, it is not possible to extrapolate accurately to man as he exists in a complex environment from the currently inadequate toxicological data derived from studies of single compounds in animals.

No data are available on the toxicity of secondary or complex atmospheric aerosols, since only a very few published reports of animal studies have appeared. The problem is highly complex because of the variability of aerosols from different urban localities and the compositional changes on collection. Evaluations of toxicity can be approached, at present, only from estimates of composition and toxicity of individual components. Using *in vitro* tests, metal oxide-coated fly ash has measurable toxicity that can be ascribed to the insoluble oxides when alveolar macrophages are exposed. The effects of soluble salts of Ni^{++} and Cd^{++} have major differences. Nickel and Cd^{++} are removed from the lung with relative rapidity, but may be stored or bound to intracellular proteins to an extent that is sufficient for accumulation on repeated short-term exposures. Two-hour exposures to both Ni^{++} (0.5 mg/m^3) and Cd^{++} (0.1 mg/m^3) aerosols impair the antibacterial defenses of the lung, leading to an increased sensitivity to airborne pathogens in mice. Ciliary beat frequency in the trachea can be decreased by Cd^{++} and Ni^{++} also. Humoral immunosuppression in mice has been reported after a 2-h exposure to 0.19 mg/m^3 $CdCl_2$ or 0.25 mg/m^3 $NiCl_2$.

It is apparent that all major types of airborne PM may contain adsorbed compounds that are mutagenic and/or carcinogenic to animals. These may contribute, to some degree, to the incidence of human cancer associated with exposure to urban air pollution.

12.6.3 Combinations of Gases and Particles

Although man is exposed to a complex mixture of gases and particles, few animal studies have been conducted with mixtures. The dissolution of SO_2 into liquid aerosols or the sorption onto solid aerosols tends to increase the potency of SO_2. The exact mechanism by which potentiation occurs is still controversial. Sodium chloride and soluble salts (manganous chloride, ferrous sulfate, or sodium orthovanadate) potentiated the effect (increased flow resistance) of a 1-h SO_2 exposure of guinea pigs. Hypothetically, these particles favored the conversion of SO_2 to H_2SO_4, thus increasing the response.

The effects of chronic exposure to a variety of mixtures of SO_2, H_2SO_4, and fly ash were examined in guinea pigs and monkeys. None of these studies showed effects on pulmonary function. Morphological changes were observed in monkeys after an 18-mo continuous exposure to 2.6 mg/m^3 (0.99 ppm) SO_2 plus 0.88 mg/m^3 H_2SO_4; but the addition of fly ash did not potentiate the response.

When dogs were exposed to SO_2 (13.4 mg/m^3, 5.1 ppm) and H_2SO_4 (0.89 mg/m^3) alone and in combination for 21 h/day for 620 days, no morphological changes were observed. Sulfur dioxide did not cause any significant changes in pulmonary function except for an increase in nitrogen washout, but H_2SO_4 caused a variety of changes that were interpreted as the development of obstructive pulmonary disease.

In another series of studies, dogs were exposed for 16 h/day for 68 mo to raw or photochemically-reacted auto exhaust, oxides of sulfur or nitrogen, or their combinations. The animals were examined periodically during exposure and at 32 to 36 mo after exposure. After 18 or 36 mo of exposure, no significant changes in pulmonary function were observed. After 61 mo, a few functional alterations were observed in dogs exposed to SO_x (1.1 mg/m^3 (0.42 ppm) SO_2, and 0.09 mg/m^3 H_2SO_4) alone and in combination with other pollutants. The animals were placed in clean air after exposure ceased, at which time the SO_x group had a variety of pulmonary functional (24 mo postexposure) and morphological alterations (32 to 36 mo postexposure). These structural changes included a loss of cilia without squamous cell metaplasia, nonciliated bronchiolar hyperplasia, and a loss of interalveolar septa in alveolar ducts. The authors hypothesized that these changes are analogous to an incipient stage of human proximal acinar (centrilobular) emphysema. Since the pulmonary function changes were progressive during the postexposure period and they were correlated with the pathology, it can be hypothesized that the morphological alterations were also progressive.

Combinations of carbon and H_2SO_4 or SO_2 were investigated also. In mice exposed for 3 h/day, 5 days/wk for up to 20 wk to a mixture of 1.4 mg/m^3 H_2SO_4 and 1.5 mg/m^3 carbon or to carbon only, morphological and immunological alterations were seen. In hamsters, a 3-h exposure to 1.1 mg/m^3 H_2SO_4 + 1.5 mg/m^3 carbon depressed ciliary beat frequency, as did H_2SO_4 alone. Alterations of both the pulmonary and systemic immune systems were found in mice at various lengths of exposure (100 h/wk up to 192 days) to 5.2 mg/m^3 (2 ppm) SO_2 and 0.56 mg/m^3 carbon, alone or in combination. Generally, carbon and carbon + SO_2 caused more extensive effects than SO_2 alone.

When the interaction of O_3 and H_2SO_4 was studied, the morphological effects of a 6-mo intermittent exposure of rats and guinea pigs to the mixture (10 mg/m^3 H_2SO_4 + 1.02 mg/m^3 (0.52 ppm) O_3) were attributed to O_3 alone. However, combined exposure to 1 mg/m^3 H_2SO_4 and 0.78 to 0.98 (0.4 to 0.5 ppm) O_3 resulted in synergistic effects on glycoprotein synthesis in the trachea and certain indices of lung biochemistry. Acute sequential exposure to first 0.196 mg/m^3 (0.1 ppm) O_3 and then 0.9 mg/m^3 H_2SO_4 caused additive effects on increased susceptibility to infectious pulmonary disease and antagonistic effects on depression of tracheal ciliary beat frequency. From these studies, the interaction of O_3 and H_2SO_4 appears quite complex and dependent on the sequence of exposure as well as on the parameter examined.

12.7 REFERENCES

Adalis, A, D.E. Gardner, F. J. Miller, and D. L. Coffin. Toxic effects of cadmium on ciliary activity using a tracheal ring model system. Environ. Res. 13:111-120, 1977.

Adalis, A., D. E. Gardner, and F. J. Miller. Cytotoxic effects of nickel on ciliated epithelium. Am. Rev. Resp. Dis. 118:347-354, 1978.

Adkins, B., Jr., J. H. Richards, and D. E. Gardner. Enhancement of experimental respiratory infection following nickel inhalation. Environ. Res. 20:33-42, 1979.

Adkins, B., Jr., G. H. Luginbuhl, and D. E. Gardner. Biochemical changes in pulmonary cells following manganese oxide inhalation. J. Toxicol. Environ. Health 6:445-454, 1980a.

Adkins, B., Jr., G. H. Luginbuhl, and D. E. Gardner. Acute exposure of laboratory mice to manganese oxide. J. Am. Ind. Hyg. Assoc. 41:494-500, 1980b.

Adkins, B., Jr., G. H. Luginbuhl, F. J. Miller, and D. E. Gardner. Increased pulmonary susceptibility to streptococcal infection following inhalation of manganese oxide. Environ Res. 23:110-120, 1980c.

Alarie, Y. Sensory irritation by airborne chemicals. CRC Crit. Rev. Toxicol. 2:299-363, 1973.

Alarie, Y., C. E. Ulrich, W. M. Busey, H. E. Swann, Jr., and H. N. MacFarland. Long-term continuous exposure of guinea pigs to sulfur dioxide. Arch. Environ. Health 21:769-777, 1970.

Alarie, Y., C. E. Ulrich, W. M. Busey, A. A. Krumm, and H. N. MacFarland. Long-term continuous exposure to sulfur dioxide in cynomolgus monkeys. Arch. Environ. Health 24:115-128, 1972.

Alarie, Y., W. M. Busey, A. A. Krumm, and C. E. Ulrich. Long-term continuous exposure to sulfuric acid mist in cynomolgus monkeys and guinea pigs. Arch. Environ. Health 27:16-24, 1973a.

Alarie, Y., R. J. Kantz II, C. E. Ulrich, A. A. Krumm, and W. M. Busey. Long-term continuous exposure to sulfur dioxide and fly ash mixtures in cynomolgus monkeys and guinea pigs. Arch. Environ. Health 27:251-253, 1973b.

Alarie, Y., C. E. Ulrich, W. M. Busey, A. A. Krumm, and H. N. MacFarland. Long-term Continuous Exposure to Sulfur Dioxide in Cynomolgus Monkeys. In: Air Pollution and the Politics of Control. MSS Information Corporation, New York, 1973c. pp. 47-60.

Alarie, Y., I. Wakisaka, and S. Oka. Sensory irritation by sulfur dioxide and chlorobenzilidene malononitrile. Environ. Physiol. Biochem. 3:53-64, 1973d.

Alarie, Y. C., A. A. Krumm, W. M. Busey, C. E. Ulrich, and R. J. Kantz. Long-term exposure to sulfur dioxide, sulfuric acid mist, fly ash, and their mixtures. Results of studies in monkeys and guinea pigs. Arch. Environ. Health 30:254-262, 1975.

Allison, A. C. Experimental methods - cell and tissue culture: effects of asbestos particles on macrophages, mesothelial cells and fibroblasts. In: Biological Effects of Asbestos. P. J. Bogorshi, V. Timbrell, J. C. Gilson, and J. C. Wagner, eds., IARC Scientific Publications No. 8, International Agency for Research on Cancer, Lyon, France. 1973. pp. 89-92.

Allison, A. C., and D. M. L. Morgan. Effects of silica, asbestos, and other particles on macrophage and neutrophil lysosomes. In: Lysosomes in Biology and Pathology, vol. 6, J. T. Dingle, P. J. Jaques, and I. H. Shaw, eds., North Holland, New York, NY, 1979. pp. 149-159.

Amdur, M. O. Effect of a combination of SO_2 and H_2SO_4 on guinea pigs. Public Health. Rep. 69:503-506, 1954.

Amdur, M. O. The respiratory response of guinea pigs to sulfuric acid mist. Arch. Ind. Health 18:407-414, 1958.

Amdur, M. O. The physiological response of guinea pigs to atmospheric pollutants. Int. J. Air Pollut. 1:170-183, 1959.

Amdur, M. O. The effect of aerosols on the response to irritant gases. In: Inhaled Particles and Vapors. C. N. Davies, ed., Pergamon Press, Oxford, England, 1961. pp. 281-294.

Amdur, M. O. The effect of high flow-resistance on the response of guinea pigs to irritants. Am. Ind. Hyg. Assoc. J. 25:564-568, 1964.

Amdur, M. O. Respiratory absorption data and SO_2 dose-response curves. Arch. Environ. Health 12:729-732, 1966.

Amdur, M. O. Toxicologic appraisal of particulate matter, oxides of sulfur, and sulfuric acid. J. Air Pollut. Control Assoc. 19:638-646, 1969.

Amdur, M. O. Aerosols formed by oxidation of sulfur dioxide. Review of their toxicology. Arch. Environ. Health 23:459-468, 1971.

Amdur, M. O. Animal studies. In: Proceedings of the Conference on Health Effects of Air Pollutants, Washington, D.C. October 3-5, 1973. A Report prepared for the Committee on Public Works, United States Senate. Serial No. 93-15, U.S. Government Printing Office, Washington, DC, 1973. pp. 175-205.

Amdur, M. O. 1974 Cummings Memorial Lecture. The long road from Donora. Am. Ind. Hyg. Assoc. J. 35:589-597, 1974.

Amdur, M. O. Toxicological Guidelines for Research on Sulfur Oxides and Particulates. In: Proceedings of the 4th Symposium on Statistics and the Environment, Washington, DC, March 3-5, 1976. American Statistical Association, Washington, DC, 1977. 48-55.

Amdur, M. O., and M. Corn. The irritant potency of zinc ammonium sulfate of different particle sizes. Am. Ind. Hyg. Assoc. J. 24:326-333, 1963.

Amdur, M. O., and J. Mead. A method for studying the mechanical properties of the lungs of unanesthetized animals. In: Proceedings of the 3rd National Air Pollution Symposium. National Air Pollution Symposium, Pasadena, CA. April, 1955. pp. 150-159.

Amdur, M. O., and J. Mead. Mechanics of respiration in unanesthetized guinea pigs. Am. J. Physiol. 192:364-368, 1958.

Amdur, M. O., and D. Underhill. The effect of various aerosols on the response of guinea pigs to sulfur dioxide. Arch. Environ. Health 16:460-468, 1968.

Amdur, M. O., and D. W. Underhill. Response of guinea pigs to a combination of sulfur dioxide and open hearth dust. J. Air Pollut. Control Assoc. 20:31-34, 1970.

Amdur, M. O., R. Z. Schulz, and P. Drinker. Toxicity of sulfuric acid mist to guinea pigs. AMA Arch. Ind. Hyg. Occup. Med. 5:318-329, 1952.

Amdur, M. O., J. Bayles, V. Ugro, and D. W. Underhill. Comparative irritant potency of sulfate salts. Environ. Res. 16:1-8, 1978a.

Amdur, M. O., M. Dubriel, and D. A. Creasia. Respiratory response of guinea pigs to low levels of sulfuric acid. Environ. Res. 15:418-423, 1978b.

Amdur, M. O., V. Ugro, and D. W. Underhill. Respiratory response of guinea pigs to ozone alone and with sulfur dioxide. Am. Ind. Hyg. Assoc. J. 39:958-961, 1978c.

American Conference of Governmental Industrial Hygenists. TLVs: Threshold Limit Values for Chemical Substances in Workroom Air Adopted by ACGIH for 1979. ACGIH, Cincinnati, OH, 1979.

Aranyi, C., F. J. Miller, S. Anders, R. Ehrlich, J. Fenters, D. E. Gardner, and M. D. Waters. Cytotoxicity of alveolar macrophages of trace metals adsorbed on fly ash. Environ. Res. 20:14-23, 1979.

Arrigoni, O. The enzymatic oxidation of sulphite in mitochondrial preparations of pea internodes. Ital. J. Biochem. 7:181-186, 1959.

Asada, K., and K. Kiso. Initiation of aerobic oxidation of sulfite by illuminated spinach chloroplasts. Eur. J. Biochem. 33:253-257, 1973.

Asahina, S., J. Andrea, A. Carmel, E. Arnold, Y. Bishop, S. Joshi, D. Coffin, and S. S. Epstein. Carcinogenicity of organic fractions of particulate pollutants collected in New York City and administered subcutaneously to infant mice. Cancer Res. 32:2263-2268, 1972.

Backstrom, H. L. J. The chain-reaction theory of negative catalysis. J. Am. Chem. Soc. 49:1460-1471, 1927.

Balchum, O. J., J. Dybicki, and G. R. Meneely. The dynamics of sulfur dioxide inhalation, absorption, distribution, and retention. Arch. Ind. Health 21:564-569, 1960.

Barry, D. H., and L. E. Mawdesley-Thomas. Effect of sulphur dioxide on the enzyme activity of the alveolar macrophage of rats. Thorax 25:612-614, 1970.

Bingham, E., E. A. Pfitzer, W. Barkley, and E. P. Radford. Alveolar macrophages: Reduced number in rats after prolonged inhalation of lead sesquioxide. Science 162:1297-1299, 1968.

Bingham, E., W. Barkley, M. Zerwas, K. Stemmer, and P. Taylor. Responses of alveolar macrophages to metals. I. Inhalation of lead and nickel. Arch. Environ. Health 25:406-414, 1972.

Boushey, H. A., M. J. Holtyman, J. R. Sheller, and J. A. Nadel. Bronchial hyperreactivity. Am. Rev. Resp. Dis. 121:389-413, 1980.

Breuninger, H. Über das physikalisch - chemische Verhalten des Nasenschleims. [Physical-chemical behavior of nasal mucus.] Arch. Ohren Nasen Kehlkopfheilkd. 184:133-138, 1964.

Brink, C., P. G. Duncan, M. Midzenski, and J. S. Douglas. Response and sensitivity of female guinea pig respiratory tissues to agonists during ontogenesis. J. Pharmacol. Exp. Ther. 215:426-433, 1980.

Brooks, A. L., R. K. Wolff, R. E. Royer, C. R. Clark, A. Sanchez, and R. O. McClellan. Biological availability and mutagic chemicals associated with diesel exhaust particles. EPA International Symposium on the Health Effects Diesel Engine Emission, Cincinnati, December, 1979. pp. 345-357.

Brune, H. F. K. Experimental results with percutaneous applications of automobile exhaust condensates in mice. Air Pollution and Cancer in Man. IARC Scientific Publications No. 16, 1977. pp. 41-48.

Camner, P., M. Lundborg, and P. Hellstrøm. Alveolar macrophages and 5 μm particles coated with different metals. Arch. Environ. Health 29:211-213, 1974.

Campbell, J. A. Cancer of skin and increase in incidence of primary tumours of lung in mice exposed to dust obtained from tarred roads. Br. J. Exp. Pathol. 15:287-294, 1934.

Campbell, J. A. Carcinogenic agents present in the atmosphere and incidence of primary lung tumours in mice. Br. J. Exp. Pathol. 20:122, 1939.

Campbell, J. A. Lung tumours in mice. Incidence as affected by inhalation of certain carcinogenic agents and some dusts. Br. Med. J. 1:217-221, 1942.

Cavender, F. L. Effects in rats and guinea pigs of six-month exposures to sulfuric acid mist, ozone, and their combination. J. Toxicol. Environ. Health 4:845-852, 1978.

Cavender, F. L., W. H. Steinhagen, C. E. Ulrich, W. M. Busey, B. Y. Cockrell, J. K. Haseman, M. D. Hogan, and R. T. Drew. Effects in rats and guinea pigs of short-term exposures to sulfuric acid mist, ozone, and their combination. J. Toxicol. Environ. Health 3:521-533, 1977.

Charles, J. M. A Mechanism for Inhaled Sulfate Initiated Bronchoconstriction. Ph.D. Thesis, Duke University, Durham, NC, 1976.

Charles, J. M., and D. B. Menzel. Ammonium and sulfate ion release of histamine from lung fragments. Arch. Environ. Health 30:314-316, 1975a.

Charles, J. M., and D. B. Menzel. Sulfate removal from the airways and histamine release in the isolated perfused rat lung. Pharmacologist 11:213, 1975b.

Charles, J. M., W. G. Anderson, and D. B. Menzel. Sulfate absorption from the airways of the isolated perfused rat lung. Toxicol. Appl. Pharmacol. 41:91-99, 1977a.

Charles, J. M., D. E. Gardner, D. L. Coffin, and D. B. Menzel. Augmentation of sulfate ion absorption from the rat lung by heavy metals. Toxicol. Appl. Pharmacol. 42:531-538, 1977b.

Clark, C. R., and Hobbs, C. H. Mutagenicity of effluents from an experimental fluidized bed coal combustor. Environ. Mutagen. 2:101-109, 1980.

Clark, C. R., and Vigil, C. L. Influence of rat lung and liver homogenates on the mutagenicity of diesel exhaust particulate extracts. Toxicol. Appl. Pharmacol. 56:110-115, 1980.

Clemo, G. R., and E. W. Miller. Tumour promotion by the neutral fraction of cigarette smoke. Br. J. Cancer 14:651-656, 1960.

Clemo, G. R., E. W. Miller, and F. C. Pybus. The carcinogenic action of city smoke. Br. J. Cancer 9:137-141, 1955.

Cockrell, B. Y., and W. M. Busey. Respiratory tract lesions in guinea pigs exposed to sulfuric acid mist. J. Toxicol. Environ. Health 4:835-844, 1978.

Cohen, H. J., and I. Fridovich. Hepatic sulfite oxidase. Purification and properties. J. Biol. Chem. 246:359-366, 1971a.

Cohen, H. J., and I. Fridovich. Hepatic sulfite oxidase. The nature and function of the heme prosthetic groups. J. Biol. Chem. 246:367-373, 1971b.

Cohen, H. J., S. Betcher-Lange, D. L. Kessler, and K. V. Rajagopalan. Hepatic sulfite oxidase. Congruency in mitochondria of prosthetic groups and activity. J. Biol. Chem. 247:7759-7766, 1972.

Cohen, H. J., R. T. Drew, J. L. Johnson, and K. V. Rajagopalan. Molecular basis of the biological function of molybdenum. The relationship between sulfite oxidase and the acute toxicity of bisulfite and SO_2. Proc. Natl. Acad. Sci. U.S.A. 70:3655-3659, 1973.

Cohen, H. J., I. Fridovich, and K. V. Rajagopalan. Hepatic sulfite oxidase. A functional role for molybdenum. J. Biol. Chem. 246:374-382, 1974.

Committee on Biologic Effects of Atmospheric Pollutants. Lead. National Academy of Sciences, Washington, DC, 1972.

Committee on Biologic Effects of Atmospheric Pollutants. Vanadium. National Academy of Sciences, Washington, DC, 1974.

Committee on Biologic Effects of Atmospheric Pollutants. Chromium. National Academy of Sciences, Washington, DC, 1974.

Committee on Medical and Biologic Effects of Environmental Pollutants. Nickel. National Academy of Sciences, Washington, DC, 1975.

Committee on Medical and Biologic Effects of Environmental Pollutants. Arsenic. National Academy of Sciences, Washington, DC, 1977.

Commoner, B., P. Madyastha, A. Bronsdon, and A. J. Vithayathil. Environmental mutagens in urban air particules. J. Toxicol. Environ. Health 4:59-77, 1978.

Corn, M., N. Kotsko, D. Stanton, W. Bell, and A. P. Thomas. Response of rats to inhaled mixture of SO_2 and SO_2 - NaCl aerosol in air. Arch. Environ. Health. 24:248-256, 1972.

Costa, D. L., and M. O. Amdur. Effect of oil mists on the irritancy of sulfur dioxide. I. Mineral oils and light lubricating oil. Am. Ind. Hyg. Assoc. J. 40:680-685, 1979a.

Costa, D. L., and M. O. Amdur. Effect of oil mists on the irritancy of sulfur dioxide. II. Motor oil. Am. Ind. Hyg. Assoc. J. 40:809-815, 1979b.

Crisp, C. E., G. L. Fisher, and J. E. Lammert. Mutagenicity of filtrates from respirable coal fly ash. Science 199:73-75, 1978.

Daisey, J. M. Organic compounds in urban aerosols. In: Aerosols: Anthropogenic and Natural Sources and Transport. T. J. Kneip and P. J. Luoy, eds., Ann. N.Y. Acad. Sci. 338:50-69, 1980.

Daisey, J. M., T. J. Kneip, I. Hawryluk and F. Mukai. Seasonal variations in the bacterial mutagenicity of airborne particulate organic matter in New York City. Environ. Sci. Technol. 14:1487-1490, 1980.

Dehnen, W., N. Pitz, and R. Tomingas. The mutagenicity of airborne particulate pollutants. Cancer Lett. 4:5-12, 1977.

Dorange, J. L., and P. Dupuy. Mise en evidence d'une action mutagene du sulfite de sodium sur la levure. [Evidence of mutogenic action of sodium sulfite on yeast.] C. R. Seances Acad. Sci., Ser. D. 274:2798-2800, 1972.

Douglas, J. S., M. W. Dennis, P. Ridgway, and A. Bouhuys. Airway constriction in guinea pigs. Interaction of histamine and autonomic drugs. J. Phamacol. Exp. Ther. 184:169-179, 1973.

Douglas, J. S., P. Ridgway, and C. Brink. Airway responses of the guinea pig *in vivo* and *in vitro*. J. Pharmacol. Exp. Ther. 202:116-124, 1977.

Drazen, J. M. Physiologic basis and interpretation of common indices of respiratory mechanical function. Environ. Health Perspect. 16:11-16, 1976.

Dukovich, M., R. E. Yasbin, S. S. Lestz, T. H. Risby, and R. B. Zweidinger. The mutagenic and SOS-inducting potential of the soluble organic fraction collected from diesel particulate emissions. Environ. Mutagen. 3:253-264, 1981.

Duran, M.; J. Korteland; F. A. Beemer; C. van der Heiden; P. K. de Bree; M. Brink; and S. K. Wadman. Variability of sulfituria: combined deficiency of sulfite oxidase and xanthine oxidase. In: International Symposium on Models for the Study of Inborn Errors of Metabolism. F. A. Hommes (ed), Elsevier/North-Holland Biomedical Press, Amsterdam, The Netherlands, 1979. p. 103.

Ehrlich, R. Interaction between environmental pollutants and respiratory infections. *In*: Proceedings of the Symposium on Experimental Models for Pulmonary Research. D. E. Gardner, E. P. C. Hu, and J. A. Graham, eds., EPA-600/9-79-022, U.S. Environmental Protection Agency, Research Triangle Park, NC, 1979. pp. 145-163.

Ehrlich, R., J. C. Findlay, and D. E. Gardner. Susceptibility to bacterial pneumonia in animals exposed to sulfates. Toxicol. Lett. 1:325-330, 1978.

Environmental Criteria and Assessment Office. Health Assessment Document for Polycyclic Organic Matter. External Review Draft No. 1, U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC, May 1978.

Environmental Criteria and Assessment Office. Health Assessment Document for Cadmium. Preprint. EPA-600/8-79-003, U.S. Environmental Protection Agency, Research Triangle Park, NC, January 1979.

Epstein, S. S., S. Joshi, J. Andrea, N. Mantel, E. Sawicki, T. Stanley, and E. C. Tabor. Carcinogenicity of organic particulate pollutants in urban air after administration of trace quantities to neonatal mice. Nature 212:1305-1307, 1966.

Epstein, S. S., E. Arnold, J. Andrea, W. Bass, and Y. Bishop. Detection of chemical mutagens by the dominant lethal assay in the mouse. Toxicol. Appl. Pharmacol. 2:288-325, 1972.

Exon, J. H., N. M. Patton, and L. D. Koller. Hexamitiasis in cadmium-exposed mice. Arch. Environ. Health 30:463-464, 1975.

Fairchild, G. A., J. Roan, and J. McCarroll. Atmospheric pollutants and the pathogenesis of viral respiratory infection. Arch. Environ. Health 25:174-182, 1972.

Fairchild, G. A., P. Kane, B. Adams, and D. Coffin. Sulfuric acid and streptococci clearance from respiratory tracts of mice. Arch. Environ. Health 30:538-545, 1975a.

Fairchild, G. A., S. Stultz, and D. C. Coffin. Sulfuric acid effect on the deposition of radioactive aerosol in the respiratory tract of guinea pigs. Am. Indust. Hyg. Assoc. J. 36:584-594, 1975b.

Falk, H. L. and P. E. Steiner. The identification of aromatic polycyclic hydrocarbons in carbon black. Cancer Res. 12:30-39, 1952.

Fenters, J. D., J. N. Bradof, C. Aranyi, K. Ketels, R. Ehrlich, and D. E. Gardner. Health effects of long-term inhalation of sulfuric acid mist - carbon particle mixtures. Environ. Res. 19:244-257, 1979.

Ferin, J., and L. J. Leach. The effect of SO_2 on lung clearance of TiO_2 particles in rats. Am. Ind. Hyg. Assoc. J. 34:260-263, 1973.

Fishbein, L. Atmospheric mutagens. I. Sulfur oxides and nitrogen oxides. Mutat. Res. 32:309-330, 1976.

Fisher, G. L., Chrisp, C. E., and O. G. Raabe. Physical factors affecting the mutagenicity of fly ash from a coal-fired power plant. Science 204:879-881, 1979.

Frank, N. R., and F. E. Speizer. SO_2 effects on the respiratory system in dogs. Changes in mechanical behavior at different levels of the respiratory system during acute exposure to the gas. Arch. Environ. Health 11:624-634, 1965.

Frank, N. R., R. E. Yoder, E. Yokoyama, and F. E. Speizer. The diffusion of $^{35}SO_2$ from tissue fluids into the lungs following exposure of dogs to $^{35}SO_2$. Health Phys. 13:31-38, 1967.

Frank, N. R., R. E. Yoder, J. D. Brain, and E. Yokoyama. SO_2 (^{35}S labeled) absorption by the nose and mouth under conditions of varying concentration and flow. Arch. Environ. Health 18:315-322, 1969.

Fraser, D. A., M. C. Battigelli, and H. M. Cole. Ciliary activity and pulmonary retention of inhaled dust in rats exposed to sulfur dioxide. J. Air Pollut. Control Assoc. 18:821-823, 1968.

Fridovich, I., and P. Handler. Xanthine oxidase. J. Biol. Chem. 233:1578-1580, 1958.

Fridovich, I., and P. Handler. Detection of free radicals in illuminated dye solutions by the initiation of sulfite oxidation. J. Biol. Chem. 235:1835-1838, 1960.

Fromageot, P., R. Vaillant, and H. Perez-Milan. Oxydation du sulfite en sulfate par la racine d'avoine. Biochim. Biophys. Acta 44:77-85, 1960.

Furst, A. and R. T. Haro. A survey of metal carcinogenesis. Prog. Exp. Tumor Res. 12:102-133, 1969.

Gardner, D. E., and J. A. Graham. Increased pulmonary disease mediated through altered bacterial defenses. In: Pulmonary Macrophage and Epithelial Cells. C. L. Sanders, R. P. Schneider, D. E. Dagle, and H. A. Ragan, eds., ERDA Symposium Series 43, Energy Research and Development Administration, Washington, DC, 1977. pp. 1-21.

Gardner, D. E., F. J. Miller, J. W. Illing, and J. M. Kirtz. Increased infectivity with exposure to ozone and sulfuric acid. Toxicol. Lett. 1:59-64, 1977a.

Gardner, D. E., F. J. Miller, J. W. Illing, and J. M. Kirtz. Alterations in bacterial defense mechanisms of the lung induced by inhalation of cadmium. Bull. Eur. Physiopathal Resp. 13:157-174, 1977b.

Gardner, D. E. Impairment of pulmonary defenses following inhalation exposure to cadmium, nickel, and manganese. J. Aerosol Sci., in press, 1981.

Giddens, W. E., and G. A. Fairchild. Effects of sulfur dioxide on the nasal mucosa of mice. Arch. Environ. Health 25:166-173, 1972.

Gilbert, E. E. Sulfonation and Related Reactions. Wiley Interscience, New York, NY, 1965. p. 125.

Gillespie, J. R. Review of the cardiovascular and pulmonary function studies on beagles exposed for 68 months to auto exhaust and other air pollutants. (In: Long-term Effects of Air Pollutants in Canine Species.) J. F. Stara, D. L. Dungworth, J. C. Orthoefer, and W. S. Tyler eds. EPA #600/8-80-014, U.S. Environmental Protection Agency, Cincinnati, Ohio, 1980, pp. 115-154.

Goff, E. U., J. R. Coombs, D. H. Fine, and T. M. Baines. Determination of n-nitrosamines from diesel engine crankcase emissions. Anal. Chem. 52:1833, 1980.

Goldring, I. P., L. Greenburg, S. S. Park, and I. M. Ratner. Pulmonary effects of sulfur dioxide exposure in the Syrian hamster. II. Combined with emphysema. Arch. Environmental Health 21:32-37, 1970.

Goldstein, B., and I. Webster. Intratracheal injection into rats of size-graded silica particles. Br. J. Ind. Med. 23:71-74, 1966.

Graham, J. A., D. E. Gardner, F. J. Miller, M. J. Daniels, and D. L. Coffin. Effect of nickel chloride on primary antibody production in the spleen. Environ. Health Perspect. 12:109-113, 1975a.

Graham, J. A., D. E. Gardner, M. D. Waters, and D. C. Coffin. Effect of trace metals on phagocytosis by alveolar macrophages. Infect. Immun. 11:1278-1283, 1975b.

Graham, J. A., F. J. Miller, M. J. Daniels, E. A. Payne, and D. E. Gardner. Influence of cadmium, nickel, and chromium on primary immunity in mice. Environ. Res. 16:77-87, 1978.

Grant, M. M., S. P. Sorokin, and J. D. Brain. Lysosomal enzyme activities in pulmonary macrophages from rabbits breathing iron oxide. Am. Rev. Resp. Dis. 120:1003-1012, 1979.

Green, G. M. The J. Burns Amberson Lecture - In defense of the lung. Am. Rev. Resp. Dis. 102:691-703, 1970.

Grose, E. C., D. E. Gardner, and F. J. Miller. Response of ciliated epithelium to ozone and sulfuric acid. Environ. Res. 22:377-385, 1980.

Grunstein, M. M., M. Hazucha, J. Sorli, and J. Milic-Emili. Effect of SO_2 on control of breathing in anesthetized cats. J. Appl. Physiol: Resp. Environ. Exercise Physiol. 43:844-851, 1977.

Gunnison, A. F., and A. W. Benton. Sulfur dioxide: Sulfite. Interaction with mammalian serum and plasma. Arch. Environ. Health 22:381-388, 1971.

Gunnison, A. F., and E. D. Palmes. Persistence of plasma S-sulfonates following exposure of rabbits to sulfite and sulfur dioxide. Toxicol. Appl. Pharmacol. 24:266-278, 1973.

Habib, M. P., P. D. Paré, and L. A. Engel. Variability of airway responses to inhaled histamine in normal subjects. J. Appl. Physiol.: Resp. Environ. Exercise Physiol. 47:51-58, 1979.

Hackney, J. D. Effects of sulfate aerosols upon cardiovascular function in squirrel monkeys. Final Report. APRAC Project CAPM-20-74, Coordinating Research Council, Inc., New York, Dec. 1, 1978.

Hadley, J. G., D. E. Gardner, D. L. Coffin, and D. B. Menzel. Inhibition of antibody mediated rosette formation by alveolar macrophages: A sensitive assay for metal toxicity. RES J. Reticuloendothel. Soc. 22:417-425, 1977.

Hammond, P. B., and R. B. Beliles. Metals. In: Toxicology: The basic science of poisons. Second edition, J. Doull, C. D. Kalassen and M. O. Amdur, (eds.). MacMillan Publishing Co., Inc. New York, NY, 1980. pp. 409-462.

Hansen, L. O., D. J. Eatough, L. Whiting, C. H. Bartholomew, C. L. Cluff, R. M. Izatt, and J. J. Christensen. Transition metal $SO_3{}^2$ complexes: A postulated mechanism for the synergistic effects of aerosols and SO_2 on the respiratory track. In: Trace Substances in Environmental Health Vol. VIII U. Montana Press, 1974. pp. 393-397.

Harkness, D. R., and S. Roth. Purification and properties of 2,3-diphosphoglyceric acid phosphatase from human erythrocytes. Biochem. Biophys. Res. Commun. 34:849-856, 1969.

Hatch, G. E., D. E. Gardner, and D. B. Menzel. Stimulation of oxidant production in alveolar macrophages by pollutants and latex particles. Environ. Res. 23:121-136, 1980.

Hayatsu, H. Bisulfite modification of nucleic acids and their constituents. Prog. Nucl. Acid Res. Mol. Biol. 16:75-124, 1976.

Hayatsu, H., and A. Miura. The mutagenic action of sodium bisulfite. Biochem. Biophys. Res. Commun. 39:156-160, 1970.

Hayon, E., A. Treinin, and J. Wilf. Electronic spectra, photochemistry, and autoxidation mechanism of the sulfite-bisulfite-pyrosulfite systems. The SO_2, SO_3, SO_4, and SO_5 radicals. J. Am. Chem. Soc. 94:47-57, 1972.

Heidelberger, C. Oncogenic transformation of cell cultures by polycyclic aromatic hydrocarbons and their derivatives. In: Polycyclic Aromatic Hydrocarbons and Cancer, Vol. 2. H. V. Gilboin and P. O. P. T'So, eds. Academic Press, New York, NY 1978.

Hemeon, W. C. L. The estimation of health hazards from air pollution. AMA Arch. Ind. Health 11:397-402, 1955.

Henderson, T. R., A. P. Li, R. E. Royer, and C. R. Clark. Increased cytotoxicity and mutagenicity of diesel fuel after reaction with NO_2. Environ. Mutagen. 3:211-220, 1981.

Heppleston, A. G. The disposal of dust in the lungs of silicotic rats. Am. J. Path. 40:493-506, 1962.

Hirsch, J. A., E. W. Swenson, and A. Wanner. Tracheal mucous transport in beagles after long-term exposure to 1 ppm sulfur dioxide. Arch. Environ. Health 30:249-253, 1975.

Holma, B., J. Lindegren, and J. M. Andersen. pH effects on ciliomotility and morphology of respiratory mucosa. Arch. Environ. Health 32:216-226, 1977.

Horvath, S. M., and L. J. Folinsbee. Interactions of Two Air Pollutants, Sulfur Dioxide and Ozone, on Lung Functions. University of California, Institute of Environmental Stress, Santa Barbara, CA, 1977.

Howell, L. G., and I. Fridovich. Sulfite: Cytochrome c oxidoreductase. J. Biol. Chem. 243:5941-5947, 1968.

Huisingh, J. L. Bioassay of particulate organic matter from ambient air. In: Application of Short-Term Bioassays in the Fractionation and Analysis of Complex Environmental Mixtures, Vol. II, Plenum Press, New York, NY, 1981.

Huisingh, J., R. Bradow, R. Jungers, L. Claxton, R. Zweidinger, S. Tejada, J. Bumgarner, F. Duffield, and M. Waters. Application of bioassay to the characterization of diesel particle emissions. Characterization of light and heavy duty diesel particle emissions. EPA-600/9-78-027, U.S. Environmental Protection Agency, Health Effects Research Laboratory and Environmental Sciences Research Laboratory, Research Triangle Park, NC, 1977.

Hyde, D., J. Orthoefer, D. Dungworth, W. Tyler, R. Carter, and H. Lum. Morphometric and morphologic evaluation of pulmonary lesions in beagle dogs chronically exposed to high ambient levels of air pollutants. Lab. Invest. 38:455-469, 1978.

Iida, S., M. Inoue, K. Kai, N. Kitamura, I. Kudo, M. Sono, T. Tsuruo, H. Hayatsu, A. Miura, and Y. Wataya. Some properties of the damage of DNA and phage 2 induced by bisulfite. Mutat. Res. 20:433-434, 1974.

International Agency for Research on Cancer. IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Man. Vol. 1, 1972, pp. 40-50.

International Agency for Research on Cancer. IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Man. Vol. 2, 1973, pp. 126-149.

International Agency for Research on Cancer. IARC Monographs on the Evaluation of Carcinogenic Risk of Chemicals to Man. Vol. 12, 1976, pp. 75-112.

Irreverre, F., S. H. Mudd, W. D. Heizer, and L. Laster. Sulfite oxidase deficiency: studies of a patient with mental retardation, dislocated ocular lenses, and abnormal urinary excretion of S-sulfo-L-cysteine, sulfite, and thiosulfate. Biochem. Med. 1:187-217, 1967.

Islam, M. S., E. Vastag, and W. T. Ulmer. Sulphur-dioxide induced bronchial hyperreactivity against acetylcholine. Int. Arch. Arbeitsmed. 29:221-232, 1972.

Kaden, D. A., R. A. Hites, and W. G. Thilly. Mutagenicity of soot and associated polycyclic aromatic hydrocarbons to Salmonella typhimurium. Cancer Res. 39:4152-4159, 1979.

Kamogawa, A., and T. Fukui. Inhibition of a-glycan phosphorylase by bisulfite competition at the phosphate binding site. Biochim. Biophys. Acta 302:158-166, 1973.

Kaplan, D., C. McJilton, and D. Luchtel. Bisulfite induced lipid oxidation. Arch. Environ. Health 30:507-509, 1975.

Katz, G. V., and S. Laskin. Pulmonary macrophage response to irritant gases. In: Air Pollution and the Lung. E. F. Aharonson, A. Ben-David, and M. A. Klingberg, eds., John Wiley and Sons, New York, NY, 1976. pp. 83-100.

Ketels, K. V., J. N. Bradof, J. D. Fenters, and R. Ehrlich. SEM studies of the respiratory tract of mice exposed to sulfuric acid mist-carbon particle mixtures. In: Scanning Electron Microscopy. Volume II, IIT Research Institute, Chicago, IL, 1977. pp. 519-526.

Kikigawa, K., and K. Iizuka. Inhibition of platelet aggregation by bisulfite-sulfite. J. Pharm. Sci. 61:1904-1907, 1972.

King, L. C., Kohan, M. J., Austin, A. C., Claxton, L. D., and Huisingh, J. L. Evaluation of the release of mutagens from diesel particles in the presence of physiological fluids. Environ. Mutagen. 3:109-121, 1981.

Klebanoff, S. J. The sulfite-activated oxidation of reduced pyridine nucleotides by peroxidase. Biochim. Biophys. Acta 48:93-103, 1961.

Kleinman, M. T., R. M. Bailey, Y. C. Chang, K. W. Clark, M. P. Jones, W. S. Linn, and J. D. Hackney. Exposures of human volunteers to a controlled atmospheric mixture of ozone, sulfur dioxide and sulfuric acid. Am. Ind. Hyg. Assoc. J. 42:61-69, 1981.

Kolber, A., T. Wolff, T. Hughes, E. Pellozzari, C. Sparacino, M. Waters, J. L. Huisingh, and L. Claxton. Collection, chemical fractionation and mutagenicity bioassay of ambient air particulate. In: Application of Short-Term Bioassays in the Fractionation and Analysis of Complex Environmental Mixtures, Vol. II, Plenum Press, New York, NY, 1981.

Koller, L. D., J. H. Exon, and J. G. Roan. Antibody suppression by cadmium. Arch. Environ. Health 30:598-601, 1975.

Kotin, P., H. L. Falk, and M. Thomas. Aromatic hydrocarbons: III. Presence in particulate phase of diesel-engine exhaust and carcinogenicity of exhaust extracts. AMA Arch. Ind. Health 11:113-120, 1955.

Kotin, P., H. L. Falk, P. Mader, and M. Thomas. Aromatic hydrocarbons. I. Presence in the Los Angeles atmosphere and the carcinogenicity of atmospheric extracts. AMA Arch. Ind. Hyg. Occup. Med. 9:153, 1954.

Kubitschek, H. E., and L. Venta. Mutagenicity of coal fly ash from electric power plant precipitators. Environ. Mutagen. 1:79-82, 1979.

Kyselá, B., D. Jiráková, R. Holusa, and V. Skoda. The influence of the size of quartz dust particles on the reaction of lung tissue. Ann. Occup. Hyg. 16:103-109, 1973.

LaBelle, C. W., J. E. Long, and E. E. Cristofano. Synergistic effects of aerosols: Particles as carriers of toxic vapors. Arch. Ind. Health 11:297-304, 1955.

Lamb, D., and L. Reid. Mitotic rates, goblet cell increase, and histochemical changes in mucus in rat bronchial epithelium during exposure to sulphur dioxide. J. Pathol. Bacteriol. 96:97-111, 1968.

Larson, T. V., D. S. Covert, R. Frank, and R. J. Charlson. Ammonia in the human airways: Neutralization of inspired acid sulfate aerosols. Science 197:161-163, 1977.

Larson, T. V., R. Frank, D. S. Covert, D. Holub, and M. S.Morgan. Measurements of respiratory ammonia and the chemical neutralization of inhaled sulfuric acid aerosol in anesthetized dogs. Am. Rev. Resp. Dis. (in press), 1982.

Laskin, S., M. Kuschner, and R. T. Drew. Studies in pulmonary carcinogenesis. In: Inhalation Carcinogenesis. AEC Symposium Series 18. M. G. Hanna, Jr., P. Nettesheim, and J. R. Gilbert, eds., U. S. Atomic Energy Commission, Oak Ridge, TN, 1970. pp. 321-351.

Laskin, S., M. Kuschner, A. Sellakumar, and G. V. Katz. Combined carcinogen-irritant animal inhalation studies. In: Air Pollution and the Lung. E. F. Aharonson, A. Ben-David, and M. A. Klingberg eds., John Wiley and Sons, New York, NY, 1976. pp. 190-213.

Last, J. A., and C. E. Cross. A new model for health effects of air pollutants: Evidence for synergistic effects of mixtures of ozone and sulfuric acid aerosols on rat lungs. J. Lab. Clin. Med. 91:328-339, 1978.

Lau, T. J., R. L. Hackett, and F. W. Sunderman. The carcinogenicity of intravenous nickel carbonyl in rats. Cancer Res. 32:2253-2258, 1972.

Lebowitz, M. D., and G. A. Fairchild. The effects of sulfur dioxide on A_2 influenza virus on pneumonia and weight reduction in mice: An analysis of stimulus-response relationships. Chem. Biol. Interact. 7:317-326, 1973.

Lee, F. S. C., W. R. Pierson, and J. Ezike. The problem of PAH degradation during filter collection of airborne particulates - an evaluation of several commonly used filter media. In: Polynuclear Aromatic Hydrocarbons: Chemical and Biological Effects. Proceedings of the Fourth International Symposium. Battelle Press, Columbus, OH, 1980.

Leikauf, G., D. B. Yeates, K. A. Wales, D. Spektor, R. E. Albert and M. Lippmann. Effects of sulfuric acid aerosol on respiratory mechanics and mucociliary particle clearance in healthy non-smoking adults. Am. Ind. Hyg. Assoc. J. 42:273-292, 1981.

Leiter, J., and M. J. Shear. Production of tumors in mice with tars from city air dusts. J. Natl. Cancer Inst. 3:167, 1942.

Leiter, J., M. B. Shimkin, and M. J. Shear. Production of subcutaneous sarcomas in mice with tars extracted from atmospheric dusts. J. Natl. Cancer Inst. 3:155-165, 1942.

Leong, K. J., H. N. MacFarland, and E. A. Sellers. Acute sulfur dioxide toxicity. Effects of histamine and histamine liberation. Arch. Environ. Health 3:66-73, 1961.

Lewis, T. R., D. E. Campbell, and T. R. Vaught, Jr. Effects on canine pulmonary function via induced NO_2 impairment, particulate interaction and subsequent SO_x. Arch. Environ. Health 18:596-601, 1969.

Lewis, T. R., W. J. Moorman, W. F. Ludmann, and K. I. Campbell. Toxicity of long-term exposure to oxides of sulfur. Arch. Environ. Health 26:16-21, 1973.

Lewis, T. R., W. J. Moorman, Y. Yang, and J. F. Stara. Long-term exposure to auto exhaust and other pollutant mixtures. Effects on pulmonary function in the beagle. Arch. Environ. Health 29:102-106, 1974.

Li, A. P. Antagonistic effects of animal sera, lung and liver cytosols, and sulfhydryl compounds on the cytotoxicity of diesel exhaust particle extracts. Toxicol. Appl. Pharmacol. 57:55-62, 1981.

Li, A. P., Brooks, A. L., and Royer, R. E. Cytotoxicity and mutagenicity of diesel exhaust soot-extracts. Proc. Annu. Meeting Environ. Mutagen Society 11:94 (Abstract), 1980.

Liber, H. L., B. M. Andon, R. A. Hites, and W. G. Thilly. Diesel soot mutation measurements in bacterial and human cells. Health Effects of Diesel Engine Emissions: Proceedings of an International Symposium, EPA-600/9-80-057a Vol. 1, pp. 402-412, 1979.

Lippmann, M., R. E. Albert, D. B. Yeates, K. Wales, and G. Leikauf. Effect of sulfuric acid mist on mucociliary bronchial clearance in healthy non-smoking humans. J. Aerosol Sci. 11:247, 1980 (abstract).

Lippmann, M., R. B. Schlesinger, G. Leikauf, D. Spektor, and R. E. Albert. Effects of sulfuric acid aerosols on respiratory tract airways. In: Inhaled Particles. V. W. H. Walton, ed., Univin Brothers Imt., Surrey, England (In Press), 1981.

Loring, S. H., J. M. Drazen, J. R. Snapper, and R. H. Ingram. Vagal and aerosol histamine interactions on airway responses in dogs. J. Appl. Physiol.: Resp. Environ. Exercise Physiol. 45:40-44, 1978.

Lyric, R. M., and I. Suzuki. Enzymes involved in the metabolism of thiosulfate by thiobacillus thioparus. Survey of enzymes and properties of sulfite: Cytochrome c oxidoreductase. Can. J. Biochem. 48:334-343, 1970.

Maigetter, R. Z., R. Ehrlich, J. D. Fenters, and D. E. Gardner. Potentiating effects of manganese dioxide on experimental respiratory infections. Environ. Res. 11:386-391, 1976.

Malanchuk, M., N. P. Barkley, and G. L. Conter. Interference of animal source ammonia with exposure chamber atmospheres containing acid particulate from automobile exhaust. J. Environ. Pathol. Toxicol. 4:256-276, 1980.

Martin, S. W., and R. A. Willoughby. Effect of sulfur dioxide on the respiratory tract of swine. J. Am. Vet. Med. Assoc. 159:1518-1522, 1971.

Marunouchi, T., and T. Mori. Studies on the sulfite-dependent ATPase of a sulfur oxidizing bacterium, thiobacillus thiooxidans. J. Biochem. 62:401-407, 1967.

Massey, V., F. Müller, R. Feldberg, M. Schuman, P. A. Sullivan, L. G. Howell, S. G. Mayhew, R. G. Matthews, and G. P. Foust. The reactivity of flavoproteins with sulfite. J. Biol. Chem. 244:3999-4005, 1969.

Matsumura, Y. The effects of ozone, nitrogen dioxide, and sulfur dioxide on the experimentally induced allergic respiratory disorder in guinea pigs. I. The effect on sensitization with albumin through the airway. Am. Rev. Resp. Dis. 102:430-437, 1970a.

Matsumura, Y. The effects of ozone, nitrogen dioxide, and sulfur dioxide on the experimentally induced allergic respiratory disorder in guinea pigs. III. The effect on the occurrence of dyspneic attacks. Am. Rev. Resp. Dis. 102:444-447, 1970b.

McCord, J. M., and I. Fridovich. Superoxide dismutase. J. Biol. Chem. 244:6049-6055, 1969a.

McCord, J. M., and I. Fridovich. The utility of superoxide dismutase in studying free radical reactions. J. Biol. Chem. 244:6056-6063, 1969b.

McDonald, S. Jr., and D. L. Woodhouse. On the nature of mouse lung adenomata, with special reference to the effects of atmospheric dust on the incidence of these tumours. J. Pathol. Bacteriol. 54:1-12, 1942.

McJilton, C., R. Frank, and R. Charlson. Role of relative humidity in the synergistic effect of a sulfur dioxide-aerosol mixture on the lung. Science 182:503-504, 1973.

McJilton, C. E., R. Frank, and R. J. Charlson. Influence of relative humidity on functional effects of an inhaled SO_2-aerosol mixture. Am. Rev. Respir. Dis. 113:163-169, 1976.

Michoud, M. C., P. D. Paré, R. Boucher, and J. C. Hogg. Airway responses to histamine and methacholine in Ascaris suum-allergic rhesus monkeys. J. Appl. Physiol.: Resp. Environ. Exercise Physiol. 45:846-851, 1978.

Miller, E. C. Some current perspectives on chemical carcinogenesis in humans and experimental animals: presidential address. Cancer Res. 38:1479-1496, 1978.

Mittler, S., and S. Nicholson. Carcinogenicity of atmospheric pollutants. Ind. Med. Surg. 26:135, 1957.

Møller, M., and I. Alfheim. Mutagenicity and PAH-analysis of airborne particulate matter. Atmos. Environ. 14:83-88, 1980.

Mossman, B. T., K. B. Adler, and J. E. Craighead. Interaction of carbon particles with tracheal epithelium in organ culture. Environ. Res. 16:110-122, 1978.

Mudd, S. H., F. Irreverre, and L. Laster. Sulfite oxidase deficiency in man: demonstration of the enzymatic defect. Science 156:1599-1602, 1967.

Mukai, F., I. Hawryluk, and R. Shapiro. The mutagenic specificity of sodium bisulfite. Biochem. Biophys. Res. Commun. 39:983-988, 1970.

Muller, F., and V. Massey. Flavin-sulfite complexes and their structures. J. Biol. Chem. 244:1007-1016, 1969.

Nadel, J. A., H. Salem, B. Tamplin, and Y. Tokiwa. Mechanism of broncho-constriction during inhalation of sulfur dioxide; reflex involving vagus nerves. Arch. Environ. Health 10:175-178, 1965a.

Nadel, J. A., H. Salem, B. Tamplin, and Y. Tokiwa. Mechanism of bronchoconstriction during inhalation of sulfur dioxide. J. Appl. Physiol. 20:164-167, 1965b.

Nadel, J. A., M. Corn, S. Zwi, J. Flesch, and P. Graff. Location and mechanism of airway constriction after inhalation of histamine aerosol and inorganic sulfate aerosol. In: Inhaled Particles and Vapours. Vol II. C. N. Davies, ed., Pergamon Press, Oxford, England 1967. p. 55-67.

Nakamura, S. Initiation of sulfite oxidation by spinach ferredoxin-NADP reductase and ferredoxin system: A model experiment on the superoxide anion radical production by metallo-flavoproteins. Biochem. Biophys. Res. Commun. 41:177-183, 1970.

National Academy of Sciences. Asbestos: The need for and feasibility of air pollution controls. National Academy of Sciences, Washington, DC, 40 pp., 1971.

National Academy of Science. Arsenic. National Academy of Sciences, Washington, DC, 1977.

National Academy of Sciences. Iron. University Park Press, Baltimore, MD, 1979a.

National Academy of Sciences. Zinc. University Park Press, Baltimore, MD, 1979b.

National Air Pollution Control Administration. Air Quality Criteria for Sulfur Oxides. AP-50, U.S. Government Printing Office, Washington, DC, 1970.

National Institute for Occupational Safety and Health. Criteria for a Recommended Standard: Occupational Exposure to Crystalline Silica. Ch. III, Biologic Effects of Exposure. DAEW (NIOSH), Pub. No. 75-120, Cincinatti, OH, 1975. pp. 15 foll.

National Research Council. Sulfur Oxides. National Academy of Sciences, Washington, DC, 1978.

National Research Council. Airborne Particles. University Park Press, Baltimore, MD, 1979.

Nulsen, A., P. G. Holt, and D. Keast. Sulfur dioxide. Acute effects on cell metabolism in vitro. IRCS Libr. Compend. 2:1464, 1974.

Office of Research and Development. Air Quality Criteria for Lead. EPA-600/8-77-017, U.S. Environmental Protection Agency, Washington, DC, December 1977.

Ohnishi, Y., K. Kachi, K. Sato, I. Tohara, H. Takeyoshi, and H. Tokiwa. Detection of mutagenic activity in automobile exhaust. Mutat. Res. 77:229-246, 1980.

Orthoefer, J. G., R. S. Bhatnagar, A. Rahman, Y. Yang, S. D. Lee, and J. F. Stara. Collagen and prolyl hydroxylase levels in lungs of beagles exposed to air pollutants. Environ. Res. 11:299-305, 1976.

Oshino, N., and B. Chance. The properties of sulfite oxidation in perfused rat liver; interaction of sulfite oxidase with the mitochondrial respiratory chain. Arch. Biochem. Biophys. 170:514-528, 1975.

Ottery, J., and I. P. Gormley. Some factors affecting the hemolytic activity of silicate minerals. Ann. Occup. Hyg. 21:131-139, 1978.

Pattle, R. E., F. Burgess, and H. Cullumbine. The effects of a cold environment and of ammonia on the toxicity of sulfuric acid mist to guinea pigs. J. Pathol. Bacteriol. 72:219-232, 1956.

Peacock, P. R., and J. B. Spence. Incidence of lung tumours in LX mice exposed to (1) free radicals; (2) SO_2. Br. J. Cancer 21:606-618, 1967.

Peiser, G. D., and S. F. Yang. Chlorophyll destruction by bisulfite-oxygen system. Plant Physiol. 60:277-281, 1977.

Pitts, J. N., D. Grosjean, J. M. Mischke, V. F. Simmons, and D. Poole. Mutagenic activity of airborne particulate organic pollutants. Toxicol. Lett. 1:65-70, 1977.

Pitts, J. N., Jr., Van Cauwenberghe, K. A., Grosjean, D., Schmid, J. P., Fitz, D. R., Belser, W. L., Jr., Knudson, G. B. and Hynds, P. M. Atmospheric reactions of polycyclic aromatic hydrocarbons: facile formation of mutagenic nitro derivatives. Science 202, p. 515-519, 1978.

Pott, F., R. Tomingas, and J. Misfeld. Tumours in mice after subcutaneous injection of automobile exhaust condensates. In: Air Pollution and Cancer in Man. IARC Scientific Publications No. 16, 1977. pp. 79-88.

Pylev, L. N. and L. M. Shabad. Some results of experimental studies in asbestos carcinogenesis. In: Biological Effects of Asbestos. IARC Scientific Publication No. 8, 1973, 99-106.

Reid, L. Evaluation of model systems for study of airway epithelium, cilia, and mucus. Arch. Intern. Med. 126:428-434, 1970.

Reiser, K. M., and J. A. Last. Silicosis and fibrogenesis: fact and artifact. Toxicol. 13:51-72, 1979.

Rigdon, R. H., and J. Neal. Tumors in mice induced by air particulate matter from a petrochemical industrial area. Texas Rep. Biol. Med. 29:109-123, 1971.

Rotilio, G., L. Calabrese, A. Finazzi Agro, and B. Mondovi. Indirect evidence for the production of superoxide anion radicals by pig kidney diamine oxidase. Biochem. Biophys. Acta 198:618-620, 1970.

Rudd, C. J. Diesel particulate extracts in cultured mammalian cells. Health Effects of Diesel Engine Emissions: Proceedings of an International Symposium, EPA-600/9-80-057a Vol. 1, pp. 385-403, 1979.

Rylander, R. Alterations of lung defense mechanisms against airborne bacteria. Arch. Environ. Health 18:551-555, 1969.

Rylander, R., M. Ohrstrom, P. A. Hellstron, and R. Bergstrom. SO_2 and particles: synergistic effects on guinea pig lungs. In: Inhaled Particles III. Vol. I. W. H. Walton, ed., Unwin Bros., Ltd., Surrey, England, 1970. pp. 535-541.

Sackner, M. A., D. Ford, R. Fernandez, J. Cipley, D. Peroz, M. Kwoka, M. Reinhardt, E. O. Michaelson, R. Schreck, and A. Wanner. Effects of sulfuric acid aerosol on cardiopulmonary functions in dogs, sheep and humans. Am. Rev. Respir. Dis. 118:497-510, 1978a.

Sackner, M. A., D. Perez, M. Brito, and R. M. Schreck. Effect of moderate duration exposures to sulfate and sulfuric acid aerosols on cardiopulmonary function of anesthetized dogs. Am. Rev. Respir. Dis. 117:257, 1978b.

Saffioti, U., F. Cefis and L. H. Kolb. A method for the experimental induction of bronchogenic carcinoma. Cancer Res. 18:104-124, 1968.

Saito, K., and D. B. Menzel. Nickel uptake and efflux from cultured type II pneumocytes. Pharmacologist 20:275, 1978.

Santodonato, J., D. K. Basu, and P. H. Howard. Health effects associated with diesel exhaust emissions: literature review and evaluation. EPA #600/1-78-063, U.S. Environmental Protection Agency, 1978. 165 pp.

Schiff, L. J., M. M. Bryne, J. D. Fenters, J. A. Graham, and D. E. Gardner. Cytotoxic effects of sulfuric acid mist, carbon particulates, and their mixtures on hamster tracheal epithelium. Environ. Res. 19:339-354, 1979.

Schlesinger, R. B., J. L. Gurman, and L. C. Chen. The production and characterization of a transition metal [FE(III)]-S(IV) Aerosol. Atmos. Environ. 14:1279-1287, 1980.

Schlesinger, R. B., M. Lippmann, and R. E. Albert. Effects of short-term exposures to sulfuric acid and ammonium sulfate aerosols upon bronchial airways function in donkeys. Am. Ind. Hyg. Assoc. J. 39:275-286, 1978.

Schlesinger, R. B., M. Halpern, R. E. Albert, and M. Lippmann. Effect of chronic inhalation of sulfuric acid mist upon mucociliary clearance from the lungs of donkeys. J. Environ. Pathol. Toxicol. 2:1351-1367, 1979.

Schneider, L. K., and C. A. Calkins. Sulfur dioxide-induced lymphocyte defects in human peripheral blood cultures. Environ. Res. 3:473-484, 1971.

Schroeder, H. A. A sensible look at air pollution by metals. Arch. Environ. Health 21:798-806, 1970.

Seelig, M. G., and E. L. Benignus. Coal smoke soot and tumors of the lung in mice. Am. J. Cancer 28:96-111, 1938.

Shahin, M. M., and F. Fournier. Suppression of mutation induction and failure to detect mutagenic activity with athabasca tar sand fractions. Mutat. Res. 58:29-34, 1978.

Shapiro, R. Genetic effects of bisulfite (sulfur dioxide). Mutat. Res. 39:149-176, 1977.

Shapiro, R., and J. M. Weisgras. Bisulfite-catalyzed transamination of cytosine and cytidine. Biochem. Biophys. Res. Commun. 40:839-843, 1970.

Shapiro, R., B. I. Cohen, and R. E. Servis. Specific deamination of RNA by sodium bisulphite. Nature 227:1047-1048, 1970a.

Shapiro, R., R. E. Servis, and M. Welcher. Reactions of uracil and cytosine derivatives with sodium bisulfate. A specific deamination method. J. Am. Chem. Soc. 92:422-424, 1970b.

Shih, V. E., I. F. Abroms, J. L. Johnson, M. Carney, R. Mandell, R. M. Robb, J. P. Cloherty, and K. V. Rajagopalan. Sulfite oxidase deficiency. Biochemical and clinical investigations of a hereditary metabolic disorder in sulfur metabolism. N. Engl. J. Med. 297:1022-1028, 1977.

Shortle, D., and D. Nathans. Local mutagenesis: A method for generating viral mutants with base substitutions in preselected regions of the viral genome. Proc. Natl. Acad. Sci. U.S.A. 75:2170-2174, 1978.

Silbaugh, S. A., R. K. Wolff, W. K. Johnson, J. L. Mauderly and C. A. Macken. Effects of sulfuric acid aerosols on pulmonary function of guinea pigs. J. Toxicol. and Environ. Health 7:339-352, 1981.

Singh, J. Biochemistry of silicosis. J. Sci. Ind. Res. 37:328-333, 1978.

Sivak, A. Overview and status of in vitro transformation. J. Assoc. Off. Anal. Chem. 62:889, 1979.

Skopek, T. R., Liber, H. L., Kaden, D. A., Hites, R. A. and Thilly, W. G. Mutation of human cells by kerosene soot. J. Natl. Cancer Inst. 63:309-312, 1979.

Snapper, J. R., J. M. Drazen, S. H. Loring, W. Schneider, and R. H. Ingram, Jr. Distribution of pulmonary responsiveness to aerosol histamine in dogs. J. Appl. Physiol.: Resp. Environ. Exercise Physiol. 44:738-742, 1978.

Spiegelman, J. R., G. D. Hanson, A. Lazarus, R. J. Bennett, M. Lippmann, and R. D. Albert. Effect of acute sulfur dioxide exposure on bronchial clearance in the donkey. Arch. Environ. Health 17:321-326, 1968.

Stara, J. F., D. L. Dungworth, J. G. Orthoefer and W. S. Tyler, eds. Long-term Effects of Air Pollutants in Canine Species. EPA-600/8-80-014, U.S. Environmental Protection Agency, Research Triangle Park, N.C., 1980.

Stenback, F., A. Ferrero, R. Montesano, and P. Shubik. Synergistic effect of ferric oxide on dimethylnitrosamine carcinogenesis in the Syrian golden hamster. Z. Krebsforsch. 79:31-38, 1973.

Stenback, F., J. Rowland, A. Sellabsumor. Carcinogenicity of benzo(a)pyrene and dusts in hamster lung (instilled intratracheally with titanium oxide, carbon and ferric oxide). Oncology 33:29-34, 1976.

Stern, A. C. Air Pollution. Vol. II. Analysis, Monitoring, and Surveying. Academic Press, New York, NY, London, England, 1968.

Stoner, G. D., M. B. Shinkin, M. C. Troxell, T. L. Thompson, and L. S. Terry. Test for carcinogenicity of metallic compounds by the pulmonary response in strain A mice. Cancer Res. 36:1744-1747, 1976.

Strandberg, L. G. SO_2 absorption in the respiratory tract. Arch. Environ. Health 9:160-166, 1964.

Summers, G. A., and J. W. Drake. Bisulfite mutagenesis in bacteriophage T4. Genetics 68:603-607, 1971.

Sunderman, F. W. Carcinogenic effects of metals. Fed. Proceedings, 37(1):40-46, 1978.

Sunderman, F. W. Metal carcinogenesis. Adv. Mod. Toxicol. 2:256-295, 1979.

Tager, J. M., and N. Rautanen. Sulphite oxidation by a plant mitochondrial system. I. Preliminary observations. Biochim. Biophys. Acta 18:111-121, 1955.

Takebe, I. Isolation and characterization of a new enzyme choline sulfatase. J. Biochem. 50:245-255, 1961.

Takino, Y., K. Sugahara, and I. Horino. Two lines of guinea pigs sensitive to chemical mediators and anaphylaxis. J. Allergy 47:247, 1971.

Tartar, H. V., and H. H. Garetson. The thermodynamic ionization constants of sulfurous acid at 25°. J. Am. Chem. Soc. 63:808, 1941.

Teranishi, K., K. Hamada, and H. Wantanabe. Mutagenicity in *Salmonella typhimurium* mutants of the benzenesoluble organic matter derived from air-borne particulate matter and its five fractions. Mutat. Res. 56:273-280, 1978.

Thompson, J. R., and D. M. Pace. Effects of SO_2 on established cell lines cultivated *in vitro*. Can. J. Biochem. Physiol. 40:207-217, 1962.

Timson, J. Action of sodium sulphite on the mitosis of human lymphocytes. Chromosomes Today 4:211-214, 1973.

Tokiwa, H., K. Morita, H. Takeyoshi, K. Takahashi, and Y. Ohnishi. Detection of mutagenic activity in particulate air pollutants. Mutat. Res. 48:237-248, 1977.

Tokiwa, H., K. Shigeji, K. Takahashi, and Y. Ohnishi. Mutagenic and chemical assay of extracts of airborne particulates. Mutat. Res. 77:99-108, 1980.

Tomori Z., and J. G. Widdicomble. Muscular, bronchomotor and cardiovascular reflexes elicited by mechanical stimulation of the respiratous tract. J. Physiol. 200:25-49, 1969.

Tuazon, P. T., and S. L. Johnson. Free radical and ionic reaction of bisulfite with reduced nicotinamide adenine dinucleotide and its analogues. Biochemistry 16:1183-1188, 1977.

Valencia, R., S. Abrahamson, P. Wagoner, and L. Mansfield. Testing for food additive-induced mutations in *Drosophila melanogaster*. Mutat. Res. 21:240-241, 1973.

Vaughan, T. R., Jr., L. F. Jennelle, and T. R. Lewis. Long-term exposure to low levels of air pollutants: Effects on pulmonary function in the beagle. Arch. Environ. Health 19:45-50, 1969.

Verrant, J. A., and Kittelson, D. B. Sampling and characterization of diesel exhaust aerosols. SEA Paper No. 770720, 1979.

Wang, Y. Y., and Wei, E. T. Detoxifying properties of liver homogenates, proteins and glutathione on diesel exhaust particulates. Proc. EPA Symp. Appl. Short-Term Bioassays Anal. Complex Environ. Mixtures, Vol. 2, 66 (Abstract) 1980.

Wang, Y. Y., S. M. Rappaport, R. F. Swayer, R. E. Talcott, and E. T. Wei. Direct-acting mutogens in automobile exhaust. Cancer Lett. 5:39-47, 1978.

Waters, M. D., D. E. Gardner, and D. L. Coffin. Cytotoxic effects of vanadium on rabbit alveolar macrophages *in vitro*. Toxcol. Appl. Pharmacol. 28:253-263, 1974.

Wattiaux-DeConinck, S., and R. Wattiaux. Subcellular distribution of sulfite cytochrome c reductase in rat liver tissue. Eur. J. Biochem. 19:552-556, 1974.

White, R., and C. Kuhn. Effects of phagocytosis of mineral dusts on elastase secretion by alveolar and peritoneal exudative macrophages. Arch. Environ. Health 35:106-109, 1980.

Widdicombe, J. G. Respiratory reflexes from the trachea and bronchi of the cat. J. Physiol. 123:55-70, 1954a.

Widdicombe, J. G. Receptors in the trachea and bronchi of the cat. J. Physiol. 123:71-104, 1954b.

Widdicombe, J. G., D. C. Kent, and J. A. Nadel. Mechanism of bronchoconstriction during inhalation of dust. J. Appl. Physiol. 17:613-616, 1962.

Williams, S. J., K. M. Holden, M. Sabransky, and D. B. Menzel. The distributional kinetics of Ni^{++} ions in the rat lung. Toxicol. Appl. Pharmacol. 55:85-93, 1980.

Wilson, D. F. The inhibition of mitochondrial respiration by bisulfite ions. Fed. Proc. Fed. Am. Soc. Exp. Biol. 27:830, 1968.

Wirth, J. J., W. P. Carney, and E. F. Wheelock. The effect of particle size on the immunodepressive properties of silica. J. Immunol. Methods 32:357-373, 1980.

Wolff, R. K., S. A. Silbaugh, D. G. Brownstein, R. L. Carpenter, and J. L. Mauderly. Toxicity of 0.4 and 0.8 μm sulfuric acid aerosols in the guinea pig. J. Toxicol. Environ. Health 5:1037-1047, 1979.

Wolff, R. K., B. A. Muggenburg, and S. A. Silbaugh. Effect of 0.3 and 0.9 μm sulfuric acid aerosols on tracheal mucous clearance in beagle dogs. Am. Rev. Resp. Dis. 123: 291-294, 1981a.

Wolff, R. K., G. M. Kanapilly, P. B. DeNee, and R. O. McClellan. Deposition of 0.1 μm chain aggregate aerosols in beagle dogs. J. Aerosol Sci. 12:119-29, 1981b.

Wynder, E. L., and D. Hoffman. A study of air pollution carcinogenesis. III. Carcinogenic activity of gasoline engine exhaust condensate. Cancer 15:103-108, 1962.

Yang, S. F. Biosynthesis of ethylene. Ethylene formation from methional by horseradish peroxidase. Arch. Biochem. Biophys. 122:481-487, 1967.

Yip, C. C., and L. D. Hadley. The iodination of tyrosine by myeloperoxidase and beef thyroids. The possible involvement of free radicals. Biochim. Biophys. Acta 122:406-412, 1966.

Yokoyama, E., R. E. Yoder, and N. R. Frank. Distribution of ^{35}S in the blood and its excretion in urine of dogs exposed to $^{35}SO_2$. Arch. Environ. Health 22:389-395, 1971.

Zarkower, A. Alterations in antibody response induced by chronic inhalation of SO_2 and carbon. Arch. Environ. Health 25:45-50, 1972.

Ziegler, I. Action of sulfite on plant malate dehydrogenase. Phytochemistry 13:2411-2416, 1974.

Ziskind, M., R. N. Jones, and H. Weill. Silicosis. Am. Rev. Resp. Dis. 113:643-665, 1976.

Zucker, M., and A. Nason. Hydroxylamine reductase from neurospora crassa. Methods Enzymol. 2:415-419, 1955.

APPENDIX 12-A

EPA Reanalysis of the Data of Peacock and Spence (1967) and Laskin et al. (1976).

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

DATE October 24, 1980

SUBJECT Analysis of the Laskin, et al and Peacock and Spence Data for Chapter 12 of the SOX/PM Document.

FROM Victor Hasselblad Vic Hasselblad
Andrew Stead V.H. for A.G.S
Biometry Division (MD-55)

TO Dr. Lester D. Grant
Director, ECAO (MD-52)

We analyzed the Laskin, et al data (p. 195) using multiple probit analysis. The model used was:

$$y_i = P(\beta_0 + \beta_1 S + \beta_2 B + \beta_3 C) + \varepsilon_i$$

where β_0 is the estimated background rate
β_1 is the increase due to SO_2(S)
β_2 is the increase due to BAP(B)
β_3 is the increase due to the combination of BAP and SO_2(C)
ε_i is the binomial random variation attributed to each animal
P is the probit (cumulative normal) function
and y_i is the response (1 means cancer, 0 means none).

The model was fitted using GLIM which is a British general linear model fitting package. The chi-square tests are really asymptotic chi-squares resulting from likelihood ratio tests computed by fitting reduced models.

Factor	Coefficient	Estimated Risk	Chi-square	P-value
Background (β_0)	-5.068	2.0×10^{-7}	-	-
SO_2 (β_1)	.449	1.9×10^{-6}	2.468	.116
BAP (β_2)	3.160	.028	2.511	.113
Combination (β_3)	3.744	.093	7.943	.005

The estimated effect of SO_2 alone is very small, but there is a hint that it increases the effect of the SO_2-BAP combination. BAP alone has a hint of an effect. The only truly significant effect is that of the SO_2-BAP combination. There is no way to test for interactions since the model comes close to "overpredicting" as it is. Compare the observed vs. predicted cancers:

Expose	Number of Animals	Number of Cancers	Predicted Cancers
A	15	0	0.000000
I	15	0	0.000007
A+C	30	1	0.846
I+C	30	2	2.168
A+CI	45	4	4.174
I+CI	46	9	8.776

A: air alone
I: SO_2 (10 ppm) alone
A+C: air plus benzo(a)pyrene (10 mg/m^3)
I+C: SO_2 (10 ppm) plus b(a)p (10 mg/m^3)
A+CI: air plus b(a)p (10 mg/m^3) and SO_2 (10 ppm)
I+CI: SO_2 (10 ppm) plus b(a)p (10 mg/m^3) and SO_2 (4 ppm)

We also analyzed SO_2 data of Peacock and Spence (p. 616) using 2X2 contingency tables. The tests were made using a one-sided Fisher's exact test which is appropriate for the small number of observed cases. The following are the results:

Group	Response	One-sided P value
Males	Primary Carcinoma	.604
Females	Primary Carcinoma	.056
Males	Adenoma	.065
Females	Adenoma	.011

These results can be summarized as marginally significant, with an indication that females are more susceptible. I do not have a test of the sex difference at this time, but I will try to forward a more sophisticated analysis next week.

cc: Dr. Judy Graham
Dr. Fred Miller
Dr. Daniel Menzel (Duke)

13. CONTROLLED HUMAN STUDIES

13.1 INTRODUCTION

Controlled human exposure studies provide a necessary bridge between epidemiology and animal toxicology data in attempting to characterize health effects induced by air pollution. In general, such studies should utilize exposure conditions that realistically simulate exposures experienced by human beings in their normal environment. However, the complexity and variability of the ambient environment is such that most controlled human exposure studies have typically been designed to evaluate initially the effects of exposure to single pollutants and later to examine effects of more complex mixtures of pollutants analogous to those present in the ambient environment.

The usually limited numbers of subjects that can be studied under controlled conditions and the associated high costs make it imperative that such studies be conducted under stringent conditions in order for their findings to be relevant to larger segments of the general population and, therefore, maximally useful for criteria development purposes. Ideally, the design of such controlled trials should include normal individuals of both sexes and all age groups, subjects especially sensitive to some particular pollutant, and individuals from populations suspected to be more generally at special risk for air pollution effects. Consideration should also be given to the activity levels of the subjects, ambient environmental conditions prevailing prior to the subjects' testing, and exposure variables that realistically simulate ambient conditions, including such factors as temperature, humidity, duration of exposure, and mode of exposure. Controlled human exposure studies also require proper experimental design, including such precautions as use of purified air and, ideally, double blind exposure procedures, especially where psychological factors may be involved as in testing suggestible asthmatics (double blind procedures, however, cannot always be complied with when certain pollutants or levels of some pollutants are being evaluated in view of associated medical risks). Furthermore, several concentrations of the pollutant(s) being tested should be employed in order to develop dose-response relationships, and comprehensive statistical treatment of the data should be carried out as well. In addition, adequate (even duplicate) pollutant monitoring equipment with documentation of quality control is needed. Proper attention should also be accorded to monitoring for potentially interfering pollutants inadvertently present or developing under certain exposure conditions. Ideally, in measuring dependent variables, not only should physiological and biochemical evidence be obtained, but subjective symptoms and/or changes in performance capability should also be assessed. Since the respiratory tract is the initial target of many air pollutants, proper and sensitive respiratory function measurements are a primary requirement and, since diurnal patterns are known to occur in many physiological systems, the time of day during which experiments are conducted should be controlled. Also, since various blood chemistry parameters and other non-respiratory system biochemical processes may be affected if pollutants (or their reaction products or substances absorbed on particles) enter the circulatory system, such biochemical effects should also be measured, as appropriate.

The above criteria should be applied to any evaluation of controlled human exposure studies. However, because of particular restraints placed on investigators, virtually no studies meet all of these ideal requirements. Nonetheless, certain useful information can be derived from many existing studies, and this chapter assesses those controlled human exposure studies most germane to understanding human health effects of sulfur dioxide, associated particulate matter, and nonsulfur particulate compounds encountered in the ambient air. It should be noted that laboratory studies utilizing man have generally been limited to the evaluation of acute exposure effects; thus the potential for chronic health effects cannot be fully evaluated based on such exposures.

13.2 SULFUR DIOXIDE

Exposure of man to sulfur dioxide has been shown to induce a number of physiological responses. Alterations in sensory system responses (such as irritation of eyes and nose changes in odor perception, and dark adaptation) have been reported. Various changes in respiratory system pulmonary functions have also been reported. The following sections attempt to characterize such physiological and pathophysiological changes, associated dose/response or exposure/effect relationships, and interacting factors that may substantially alter such relationships (e.g., alterations in respiratory tract SO_2-deposition patterns are dependent upon exercise levels or the health status of test subjects as noted in Chapter 11).

13.2.1 Subjective Reports

The perception of odor and the sensation of irritation in the eyes, nose, throat, or other parts of the body are difficult to measure precisely. Thus subjects may be observed for qualitative changes (coughing, rhinorrhea, lacrimation) or asked to report whether they detect something in the air they are breathing. Several studies have used such subjective reports as an indication of the effects of SO_2 on human subjects.

A number of early investigators exposed themselves to high concentrations of SO_2 (>500 ppm [1310 mg/m^3]) and experienced coughing, irritation of the eyes and nose, and difficulty in breathing (e.g., Ogata, 1884; Yamada, 1905; Kisskalt, 1904). In the course of investigating the effects of SO_2 on industrial workers, Lehman (1893) and his associates experienced nasal irritation during exposures of 10 to 15 minutes to 6.5 ppm (17 mg/m^3) SO_2. Holmes et al. (1915 -- cited by Greenwald, 1954) carried out an extensive study of 60 subjects, 28 of whom were unaccustomed to breathing SO_2, and 32 of whom were familiar with it. All of the subjects already familiar with the gas seemed to detect it (either as SO_2 or as "something foreign") at 3 ppm (7.9 mg/m^3). But only 10 of 28 unaccustomed subjects detected something in the air at 3 ppm (7.9 mg/m^3) SO_2. Few subjects found momentary whiffs of 5 ppm (13.1 mg/m^3) disagreeable, although "long-continued breathing of air containing slightly more than 5 parts per million would probably cause discomfort to most people..." (Holmes et al., 1915). Amdur et al. (1953) noted that during exposure to 1 to 2 ppm (2.6 to 5.2 mg/m^3) their subjects could not usually detect the odor of SO_2; even at 5 ppm (13.1 mg/m^3) most subjects could not smell the gas, although they did complain of dryness in the throat. One subject, however, objected so strongly to the odor of 5 ppm (13.1 mg/m^3) SO_2 that exposure was terminated. Above 5 ppm (13.1 mg/m^3) the odor was definitely detected by all subjects. In numerous more recent studies, subjects have also been asked to report their subjective experiences (e.g., Greenwald, 1954; Tomono, 1961; Frank et al., 1962; Toyama and Nakamura, 1964; Speizer and Frank, 1966a,b; Melville, 1970; Weir and Bromberg, 1972, 1973; Lawther et al., 1975; Horvath and Folinsbee, 1977), and the results seem to be quite variable at exposures less than 5 ppm SO_2.

The significance of such subjective reports is unclear at present. This is especially true in light of findings by Frank et al. (1962) showing that subjective reports are in some situations an unreliable indicator of physiological changes, since coughing and a sense of throat irritation tended to subside in their subjects after a few minutes while other changes in respiratory functions were still maximal.

13.2.2 Sensory Effects

Among the physiological functions affected by exposure to SO_2 are certain sensory processes. Studies investigating sensory effects not only evaluated odor threshold for detection of SO_2 but also SO_2 effects on the sensitivity of the dark-adapted eye and interruption of the alpha (α) rhythm in electroencephalograms (see Table 13-1). Most of these investigations have been summarized by Ryazanov (1962).

13.2.2.1 Odor Perception Threshold--In the Russian studies, odor threshold is typically determined in a well-ventilated chamber containing two orifices from which emerge two small streams of gas, one being very pure air and the other other being a stream of the test gas. The subject sits in front of the apparatus, sniffs both orifices, and points out the odorous one. This experiment is repeated with the same concentration of test gas over a period of several days. The experiment is performed with sequentially reduced concentrations until the subject, in the majority of instances, denies the presence of an odor or gives erroneous answers. The threshold concentration for the most sensitive subject in a group of volunteers is defined as the threshold for odor perception.

Using the two-orifice apparatus described above, Dubrovskaya (1957) conducted sulfur dioxide odor perception threshold tests on 12 subjects. Sulfur dioxide concentrations of 0.5 mg/m^3 to 13 mg/m^3 (0.17 ppm to 4.6 ppm) were used in 530 threshold determinations. Six test subjects sensed the odor of sulfur dioxide in the range 2.6 mg/m^3 to 3.0 mg/m^3 (1.0 to 1.1 ppm); four subjects sensed the odor in the range 1.6 mg/m^3 to 2.0 mg/m^3 (0.6 to 0.8 ppm); one sensed the odor in the range 2.1 mg/m^3 to 2.5 mg/m^3 (0.8 to 1.0 ppm); and one sensed the odor in the range 3.1 mg/m^3 to 3.6 mg/m^3 (1.2 to 1.4 ppm). Thus, the average sulfur dioxide odor threshold concentration was 0.8 ppm to 1 ppm (~2.3 mg/m^3 to ~2.9 mg/m^3), and for the more sensitive of these persons it was 0.5 ppm to 0.7 ppm (~1.5 mg/m^3 to ~2.0 mg/m^3). It should be noted, however, that most of the subjects were of an age at which odor perception was presumed to be most sensitive.

TABLE 13-1. SENSORY EFFECTS OF SO_2

Concentration SO_2 (ppm)	Exposure mins.	Effects	Reference
400	120	Dyspnea	Ogata, 1884
6.5	10 - 15	Nasal irritation	Lehman, 1893
140, 210, 240	30	Marked nasal irritation, sneezing	Yamada, 1905
210, 240	30	Eye irritation, lacrimation	Yamada, 1905
1, 2, 5	--	All subjects detect odor above 5 ppm	Amdur et al., 1953
3, 5, 5 plus	--	Discomfort to all subjects exposed to 5 plus. Some noted disagreeable odor at 5 ppm.	Holmes, 1915 (see Greenwald, 1954)
0.17 - 4.6	--	Average SO_2 odor threshold was 0.8 - 1.0 ppm	Dubrovskaya, 1957
	--	Positive recognition of SO_2 was 0.47 ppm	Arthur D. Little, Inc., 1968
0.34 - 6.9	15	Light sensitivity increased at 0.34 - 0.63 ppm and above	Dubrovskaya, 1957
0.23	--	Ocular sensitivity to light increased at SO_2 levels of 0.23 ppm and above	Shalamberidze, 1967
0.2 - 1.7	0.33	Attenuation of α-waves at levels above 0.2 ppm	Bushtueva, 1962
1 - 10	--	Organoleptic effects at levels 2 ppm and above	Greenwald, 1954

[a] 0.1 ppm $SO_2 \cong 262\ \mu g/m^3$ 1.0 ppm $\cong 2620\ \mu g/m^3$ 10 ppm $\cong 26,200\ \mu g/m^3$

0.5 ppm $SO_2 \cong 1310\ \mu g/m^3$ 5.0 ppm $\cong 13,100\ \mu g/m^3$ 50 ppm $\cong 131,000\ \mu g/m^3$

Studies of sulfur dioxide odor thresholds (1968) conducted for the Manufacturing Chemists' Association in the United States gave somewhat lower values than those cited above (Arthur D. Little, Inc., 1968). The concentration at which first one-half and then all of the panel members could positively recognize the odor was reported to be 0.47 ppm (1.3 mg/m^3). The details of the test procedure are thoroughly discussed in the report, but one important aspect is reiterated as a reminder that odor thresholds usually represent values derived under ideally suited conditions with trained individuals. That is, under ideal conditions, the investigators (who were highly qualified to judge on the basis of substantial experience with consumer evaluation of known flavor and odor situations) derived threshold values lower than those which would be recognized by the majority of people under ordinary atmospheric conditions. This does not mean that normal individuals exposed to sulfur dioxide under ideal test conditions could not perceive the 0.47 ppm (1.2 mg/m^3) level indicated. However, because of background odor and lack of awareness or concern with ambient odor conditions, such individuals would probably be less responsive to this low concentration in everyday situations.

13.2.2.2 Sensitivity of the Dark-Adapted Eye--The sensitivity of the eye to light while a subject is in darkness increases with time. Several investigations have been made of the effects of inhalation of sulfur dioxide on this sensitivity. Typically, measurements of a subject's normal sensitivity are taken in a dark, well-ventilated chamber in complete silence. (Sudden stimuli, including noise, may affect the subject's response.) Each subject is tested once daily following preliminary stimulation at a high light level. Light sensitivity is measured at 5-minute or 10-minute intervals, and a curve of increasing sensitivity to light is established from measurements taken over a period of 7 to 10 days.

Dubrovskaya (1957) studied the effect of inhaling sulfur dioxide in concentrations from 0.96 mg/m^3 to 19.2 mg/m^3 (0.4 to 7.3 ppm) for 15 minutes before measuring light sensitivity during dark adaptation. She reported that light sensitivity was increased by sulfur dioxide concentrations of 0.96 mg/m^3 to 1.8 mg/m^3 (0.34 ppm to 0.63 ppm), that the increase in sensitivity reached a maximum at concentrations of 3.6 mg/m^3 to 4.8 mg/m^3 (1.3 ppm to 1.7 ppm), and that further increases in the sulfur dioxide concentration resulted in progressive lowering of eye sensitivity to light until, at 19.2 mg/m^3 (7.3 ppm), the sensitivity was identical with that of the unexposed subject.

In exposures during light adaptation, sulfur dioxide concentrations of 0.6 mg/m^3 to 7.2 mg/m^3 (0.23 ppm to 2.8 ppm) caused slight increases in eye sensitivity. Maximum sensitivity was attained at 1.5 mg/m^3 (0.52 ppm); at higher concentrations the increased sensitivity began to abate. Two human subjects were used in these experiments. The odor threshold was between 2.5 mg/m^3 and 3.0 mg/m^3 (1.0 and 1.1 ppm) for one subject and between 3.0 mg/m^3 and 3.6 mg/m^3 (1.1 and 1.4 ppm) for the other, so that changes in sensitivity to light during dark adaptations were caused by sulfur dioxide concentrations below the odor threshold.

Shalamberidze (1967) investigated the effects of SO_2 and NO_2, singly and in combination, on visual light sensitivity as determined by measures of dark adaptation. According to this report, SO_2 concentrations of 0.6 mg/m^3 (0.23 ppm) and higher caused "a considerable increase in the ocular sensitivity to light" (Shalamberidze, 1967). So few details on methods or results were presented, however, that this report cannot be accepted without reservations.

13.2.2.3 Interruption of Alpha Rhythm--The electroencephalogram is a composite record of the electrical activity of the brain recorded as the difference in electrical potential between two points on the head. In the resting adult, the electroencephalogram characteristically shows a fairly uniform pattern of electrical frequency from 8 to 12 Hz (alpha-rhythms) in the posterior head regions. Variations occur with age, the state of wakefulness and attentiveness, or as a result of incoming sensory stimuli from exteroceptive or interoceptive receptors. The dominant frequency (α) is inhibited or attenuated by eye opening and by mental activity.

Subjects with well-defined α-rhythms studied in a silent and electrically shielded chamber show a temporary attenuation of the α-rhythm each time they are given a light signal. When the light is excluded, the α-rhythm returns to normal. A concentration of test gas is determined which is so low that by itself it does not cause attenuation of the α-rhythm. A subject breathes the gas at this concentration, and then he receives the light signal. After exposure to this sequence (gas then light) several times (5 to 30 times in 1 day), a subject will show attenuation before he receives the light signal; that is, he responds to the unperceived odor. The unperceived odor thus becomes the conditioning stimulus and brings about the so-called conditioned electrocortical reflex.

Bushtueva (1962) reported that 20-second exposures of six human subjects to sulfur dioxide concentrations from 0.9 mg/m^3 to 3 mg/m^3 (~0.3 ppm to ~1.0 ppm) produced attenuation of the α-wave lasting 2 to 6 seconds; at concentrations of 3.0 mg/m^3 to 5.0 mg/m^3 (~1.0 ppm to 1.7 ppm) attenuation lasted throughout the 20-second exposure. Exposures to 0.6 mg/m^3 (~0.2 ppm) did not cause attenuation of the α-wave. The threshold for attenuation of the α-wave was the same as the odor theshold or the threshold of irritation of the respiratory tract. Bushtueva further demonstrated that electrocortical conditioned reflexes could be developed with sulfur dioxide at 0.6 mg/m^3 (~0.2 ppm) but not with lesser concentrations of the mixture.

13.2.3 Respiratory and Related Effects

13.2.3.1 Respiratory Function--A number of controlled human exposure studies have documented various respiratory and cardiovascular effects deriving from exposure to SO_2, as summarized in Table 13-2. (See Chapters 11 and 12 for further information concerning respiratory deposition and effects of SO_2.) One of the first clinical studies examining the effects on healthy adults of inhaling SO_2 was reported by Amdur et al. (1953). They had 14 resting subjects breathing SO_2 for 10 minutes through a face mask in concentrations ranging from 1 to 8 ppm (2.6 to 21.0 mg/m^3). Pulse rate and respiration rate increased and tidal volume decreased during exposure to as little as 1 ppm SO_2. Several investigators attempted to replicate the findings of Amdur et al. (1953), including Lawther (1955), and Frank et al. (1962). None was able to find consistent respiratory or cardiovascular effects in healthy adults at SO_2 levels below 5 ppm (13.1 mg/m^3). Nevertheless, these and other studies have documented a variety of subjective and physiological effects under various conditions of exposure to SO_2.

13-2. RESPIRATORY EFFECTS OF SO_2

Concentration SO_2 (ppm)[a]	Duration of exposure (mins)	Number of subjects	Oral (O) or nasal (N) exposure	Rest (R) or exercise (E)[*]	Effects[b]	Reference
HEALTHY ADULT SUBJECTS AT REST						
10,15,25,50	60	1	-	-	Mucociliary activity decreased at higher conc. (>15 ppm SO_2)	Cralley, 1942
9 - 60	5	10	Mask	R	Airway resistance increased	Nakamura, 1964
5, 10 20	10 10	18 6	O, N O	R R	No changes in pulse rate, resp. rate or tidal vol. (5, 10 ppm). Bronchospasm in two subjects at 10 ppm	Lawther, 1955
1 - 80	10 - 60	8 - 12	Mask, chamber N	R	Bronchoconstriction above 5 ppm	Sim and Pattle, 1957
1 - 45	10	46	Mask	R	Decreased peak flow, decreased expiratory capacity at ≥ 1.6 ppm	Tomono, 1961
1 - 8	10	14	Mask	R	Pulse and respiratory rates increased; tidal volume rate decreased at ≥1.0 ppm	Amdur et al., 1953
1(1-2), 5(4-7), 13(10-16)	10 - 30	11	O[*]	R	Pulmonary flow resistance for groups increased 39% at 5 ppm and 72% at 13 ppm. At 1 ppm, one subject had 7% increase in flow resistance, another a 23% decrease	Frank et al., 1962
1(1-2), 5(4-6), 15(14-17)	30	12	O[*]	R	Increases in Rl (pulmonary flow resistance) at ≥5 ppm SO_2	Frank et al., 1964
4 - 6	10	7	O	R	Airway conductance decreased 39%. Blocked by atropine	Nadel et al., 1965
4 - 5	10	5	O[*]	R	Increased respiratory and inspiratory resistance	Abe, 1967

[*] Mouthpiece

TABLE 13-2. (continued)

Concentration SO_2 (ppm)	Duration of exposure (mins)	Number of subjects	Oral (O) or nasal (N) exposure	Rest (R) or exercise (E)	Effects	Reference
15, 28	10	8	O,N	R	Pulmonary flow resistance increased less with N breathing	Speizer and Frank, 1966a
1.0, 5.0 and 25.0	2-4 hr/d	15	Chamber (N)	R	Significant decreases in expiratory flow and $FEV_{1.0}$ at 25 ppm. Decreased nasal mucus flworate at ≥ 5 ppm. Responses greater after 4 hr than after 2 hr	Andersen et al., 1974
5	270 hr	16 controls 16 exposed	Chamber (N)	R	50% decrease in nasal mucus flowrate but number of colds similar in both groups	Andersen et al., 1977
5	120	9	O	R	No effect on mucus transport	Wolff et al., 1975a
5 - 30 1 3	10 60/DB DB	10 13/12 17	CO_2 stimulus (O)* Chamber N/O O*	R R R	For group as whole (12 subjects) small but significant (14%) increase in SR_{aw} following 25 DB by air alone and 26% increase after 25 DB SO_2 at 1 ppm; but no changes detected after normal quiet breathing of 1-3 ppm SO_2	Lawther, 1975
2.5, 5.0, 10.0	10	15	O*, N	R	Greater percentage decrease in in SG_{aw} with O breathing at all concentrations	Melville, 1970
0.5, 1.0, 5.0	15	9 5	O* O, N*	R R	Decreases in $MEF_{50\% VC}$ for group were sig. at 1 and 5 ppm SO_2; at 5 ppm, decreases for N not sig. different from O breathing	Snell and Luchsinger, 1969
1.1 - 3.6	30	10	O*	R	Deep breathing produced no effects	Burton et al., 1969
0.50	180	40	Chamber (O) Nose clips	R	No pulmonary effects seen	Jaeger et al., 1979

*Mouthpiece
DB = deep breaths

TABLE 13-2. (continued)

Concentration SO_2 (ppm)	Duration of exposure (mins)	Number of subjects	Oral (O) or nasal (N) exposure	Rest (R) or exercise (E)	Effects	Reference
EXERCISING HEALTHY ADULTS						
5.0	120	10	O	E	Increased tracheobronchial clearance	Wolff et al., 1975b
5.0	120	11	Chamber	E	Insignificant changes in airway resistance and arterial PO_2	von Nieding et al., 1979
5.0	120	10	Chamber (O)	E	MMFR decreased 8.5%; increased tracheobronchial clearance	Newhouse et al., 1978
5.0 3.0 1.0 0.5	3 3 3 3	10 8+9 10+8 5	O O O O	R R + E R + E R	Light exercise potentiates effect of SO_2. $MEF_{40\%}$ decreased at 3 ppm and above	Kreisman et al., 1976
0.75	120	4	Chamber	E	Decrease in MMFR, FVC, $FEV_{1.0}$ (~8-10%) and 20% in $MEFR_{50\%}$	Bates and Hazucha, 1973
0.75	120	15 controls 16 exposed	Chamber	E	Significantly elevated Raw and trend toward decreased FEF_{50} and FEV/FVC after SO_2 exposure during heavy exercise	Stacy et al., 1981
0.50	120	24	Chamber	E	No pulmonary effects seen with 0.50 ppm SO_2 + 0.5 ppm NO_2	Linn et al., 1980
0.40	120	9	Chamber	E	No pulmonary effects	Horvath and Folinsbee, 1977; Bedi et al., 1979
0.40	120	11	Chamber	E	No pulmonary effects seen with 0.4 ppm SO_2 alone	Bedi et al., 1981
0.37	120	8	Chamber	E	No pulmonary effects	Bates and Hazucha, 1973; Hazucha and Bates, 1975
0.37	120	4-12	Chamber	E	No pulmonary effects	Bell et al., 1977b
RESPIRATORY DISEASE SUBJECTS						
0.3, 1.0 and 3.0	96 - 120 hr	12 (normal) 7 (COPD)	Chamber	R	No difference in response between groups. Slight decrease in pulmonary compliance but of questionable significance	Weir and Bromberg, 1972
7.7 0.3-4	6 d 6-7 d	32 normals 27 subjects w/obstrutive resp. disease	Chamber (N) Chamber (N)	R R	No significant changes in airway resistance or other effects in health subjects or patients	Reichel, 1972

TABLE 13-2. (continued)

Concentration SO_2 (ppm)	Duration of exposure (mins)	Number of subjects	Oral (O) or nasal (N) exposure	Rest (R) or exercise (E)	Effects	Reference
ASTHMATIC SUBJECTS						
1, 3, 5	10	7 normals 7 atopics 7 asthmatics	O*	R	SR_{aw} increased significantly at all conc for asthmatics; only at 5 ppm for normals and atopic subjects. Some asthmatics exhibited marked dyspnea requiring bronchodilation therapy.	Sheppard et al., 1980
1.0 0.1, 0.25, 0.5	5 10	6 asthmatics 7 asthmatics	O*	E	SR_{aw} significantly increased in the asthmatic group at 0.5 and 0.25 ppm SO_2 and at 0.1 ppm in the two most responsive subjects. At 0.5 ppm three asthmatics developed wheezing and shortness of breath.	Sheppard et al., 1981
0.50	180	40 (asthmatics)	Chamber (O) Nose clips	R	MMFR significantly decreased 2.7%; recovery within 30 min.	Jaeger et al., 1979
0.5	10	5 asthmatics	O*	E	Specific airway resistance (SR_{aw}) increases were observed over exercise baseline rates for 80% of the subjects.	Linn et al., 1982
0.25, 0.5	60	24 asthmatics	Chamber	E	No statistically significant changes in forced vital capacity (FVC) or specific airway resistance (SR_{aw})	Linn et al., 1982
0.30	120	19 (asthmatics)	Chamber	E	No pulmonary effects seen with 0.3 ppm SO_2 and 0.5 ppm NO_2 exposure compared to exercise basline	Linn et al., 1980

[a] 0.1 ppm SO_2 ≅ 262 μg/m³ 1.0 ppm ≅ 2620 μg/m³ 10 ppm ≅ 26,200 μg/m³

0.5 ppm SO_2 ≅ 1310 μg/m³ 5.0 ppm ≅ 13,100 μg/m³ 50 ppm ≅ 131,000 μg/m³

[b] Significant increase or decreases noted here refer to "statistically significant" effects, independent of whether the observed effects are "medically significant" or not.

[c] Chronic obstructed pulmonary disease.

*Mouthpiece DB = deep breaths

Sim and Pattle (1957) performed extensive clinical studies over a 10-month period on an unspecified number (8 to 12) of "healthy males aged 18 to 45." Sulfur dioxide was administered either by face mask at concentrations ranging from 1.34 to 80 ppm (3.51 to 210 mg/m^3) for 10 minutes or in an inhalation chamber at concentrations of 1.0 to 23.1 ppm (2.6 to 60.5 mg/m^3) for 60 minutes. Regardless of exposure route, the only notable effects of SO_2 were said to be bronchoconstriction (increased resistance to air flow) at concentrations $\geq$ 5 ppm (13.1 mg/m^3) SO_2 and high-pitched chest rales at exposure $\geq$ 49 ppm (128 mg/m^3) SO_2. They also reported that when ammonia (no value given) was also present in the chamber (9.9 ppm [26 mg/m^3] SO_2), the subjective impressions of bronchoconstriction disappeared.

Frank et al. (1962) examined the effects of acute (10 to 30 minute) exposures to SO_2 via mouth in 11 healthy adult subjects. Each subject received approximately 1, 5, and 13 ppm (2.6, 13 and 34 mg/m^3) of the gas in separate exposures at least 1 month apart. The only statistically significant effects were a 39-percent increase ($p < 0.01$) in pulmonary flow resistance at 5 ppm (13 mg/m^3) and a 72 percent increase ($p < 0.001$) at 13 ppm (34 mg/m^3). Only one subject showed a statistically significant increase with 1 ppm (2.6 mg/m^3) SO_2 concentration; his control resistance was the highest encountered. The recovery of some subjects was complete within a few minutes. As in Sim and Pattle's study (1957), other cardiovascular or pulmonary measures did not show any statistically significant effects.

Tomono (1961) tested 46 men for the effects of SO_2 on their pulmonary physiology. The subjects inhaled 1 to 45 ppm (2.6 to 118 mg/m^3) SO_2 through face masks for 10 minutes. Decreases in expiratory capacity and peak flowrate were proportional to the concentration of SO_2. Such effects were detected at a concentration as low as 1.6 ppm (4.2 mg/m^3). Slight increases in pulse and respiration rates were observed in about 10 percent of the subjects but were not proportional to SO_2 exposures. Nakamura (1964) exposed 10 adult subjects, each to a different concentration of SO_2 (9 to 60 ppm [23.6 to 157 mg/m^3]) for 5 minutes. Airway resistance increased an average of 27 percent. Since each subject was exposed to only one concentration of SO_2 and there was considerable variability in response to the different concentrations, the significance of those isolated findings may be questioned. No statistically significant correlation between dosage and response was discovered. For example, one subject had a 17-percent increase after exposure to 9 ppm (23.6 mg/m^3), another 9 percent after exposure to 16 ppm (41.9 mg/m^3), another 75 percent after exposure to 47 ppm (123 mg/m^3), and another 22 percent after exposure to 57 ppm (149 mg/m^3).

Snell and Luchsinger (1969) also found statistically significant decreases in pulmonary function consequent to SO_2 exposure of healthy adults. Nine subjects inhaled SO_2 through a mouthpiece at concentrations of 0.5, 1.0, and 5 ppm (1.3, 2.6 and 13.0 mg/m^3) for 15 minutes each, with 15-minute control periods interspersed. Maximum expiratory flow ($MEF_{50\%\ VC}$) was significantly lower after exposure to 1 ppm (2.6 mg/m^3) SO_2 ($p < 0.02$) as well as 5 ppm (13.0 mg/m^3) ($p < 0.01$). Reichel (1972) exposed 32 normal subjects in a chamber to 7.7 ppm (20.2

mg/m^3) SO_2 continuously for 6 days. Intrathoracic gas volume and intrabronchial flow resistance were not altered consequent to any of the 6 days of exposure. Airway resistance was measured in 16 of these subjects after inhalation of 3 percent acetylcholine chloride solution. The sensitivity response to this challenge was not altered as a consequence of the exposure to SO_2. Jaeger et al. (1979) exposed 40 normal non-smoker subjects for 3 hours to 0.5 ppm (1.3 mg/m^3) SO_2. Oral inhalation was forced by having the subjects wear nose clips. These resting subjects were also studied during exposure to ambient air having an average SO_2 content of 0.005 ppm (13.1 μg/m^3). Three pulmonary function tests (VC, FEV_1 and MMFR) were performed at intervals during the exposure, and a more intensive series of tests was made prior to and after the exposure. No pulmonary changes were observed, although one subject (a probable asthmatic) complained of wheezing during the night following the SO_2 exposure; these symptoms, however, are not clearly attributable to the SO_2 exposure experienced earlier in the day.

Nadel et al. (1965) helped to elucidate the mechanism of bronchoconstriction resulting from SO_2 exposure. They exposed seven subjects to 4 to 6 ppm (10.5 to 15.7 mg/m^3) SO_2 for 10 minutes via mouth in a closed plethysmograph. The mean decrease in specific airway conductance was 39 percent ($p < 0.001$). After subcutaneous injection of atropine (1.2-1.8 mg), inhalation of SO_2 resulted in only a three-percent ($p > 0.02$) decrement in specific airway conductance/thoracic gas volume. However, atropine did not affect the coughing or sensation of irritation in the pharynx or substernal area. From this and other evidence, Nadel et al. concluded that the bronchoconstriction induced by SO_2 depends on changes in smooth muscle tone mediated by parasympathetic motor pathways. Thus, when sensory receptors in the tracheobronchial region are irritated by a substance such as SO_2, a reflexive bronchospasm may be triggered.

13.2.3.2 Water Solubility--An important point to note is that, because of its high solubility in water, SO_2 is readily absorbed when it comes in contact with the moist surfaces of the nose and upper respiratory passages (Frank et al., 1973). This has a number of important implications for the analysis of the effects of SO_2 on respiratory functions. These considerations will be illustrated in the following sections (see also Chapter 11).

13.2.3.3 Nasal Versus Oral Exposure--Several studies have demonstrated notable differences in SO_2 concentrations necessary to elicit pulmonary effects depending upon the route (i.e., via nose or mouth) of exposure to SO_2. Speizer and Frank (1966a), for example, compared the effects of SO_2 (10-minute exposures at 15 and 28 ppm [39.3 and 73.4 mg/m^3]) in eight subjects breathing the gas either by nose or by mouth. The subjects coughed less and reported less irritation of the throat and chest when breathing through their noses. Also, pulmonary flow resistance increased less during nasal exposure than during oral exposure.

A second study by the same investigators (Speizer and Frank, 1966b) refined their analysis of these effects, using seven subjects and a specially designed face mask. Air was sampled at various points, including: (1) within the face mask before being inspired, (2) within the subject's nose, and (3) within the subject's oropharynx. Exposures lasted 25 to 30 minutes. The average concentration of SO_2 within the mask was 16.1 ppm (42.2 mg/m^3); within the oropharynx the concentration was too low for the investigators' equipment to measure. Thus, essentially all of the SO_2 (90 to 99 percent) in the inspired air was removed by the nose. Similar results were obtained by Andersen et al. (1974) in a study described in detail later.

Melville (1970) also compared oral and nasal routes of administration. He used 15 subjects and exposed them (for 10 minutes) sequentially to 2.5, 5, and 10 ppm (6.6, 13.1, and 26.2 mg/m^3) SO_2. More SO_2 was removed per minute with nose breathing than with mouth breathing. There was a clear dose-dependent response reflected in measures of the subjects' specific airway conductance (SG_{aw}, not typical panting procedure). As the SO_2 concentration increased, SG_{aw} decreased ($p < 0.05$). This was true regardless of administration route, but at 2.5 ppm (6.6 mg/m^3) SO_2 the average decrease under oral administration was greater (in 80 percent of subjects) than the decrease under nasal administration ($p < 0.05$). During a 1-hour exposure to 5 ppm (13.1 mg/m^3) SO_2, no statistically significant difference was observed (49 subjects) in SG_{aw} decreases after 5-minutes of mouth breathing in comparison to marked SG_{aw} decreases after nasal breathing and no further decrements in SG_{aw} beyond the levels reached after 5 minutes exposure were observed with continuation of exposure up to 1 hour. During mouth exposures, some subjects coughed at the onset of SO_2 exposure and complained of burning sensations in the throat and substernal chest pains. Three returned a week later complaining of substernal pains and bronchial infections.

Snell and Luchsinger (1969) also examined the differences between nasal and oral exposure using SO_2 at 5 ppm (13.1 mg/m^3). Five subjects' average maximum expiratory flow ($MEF_{50\% VC}$) was 10 percent lower following oral exposure than following nasal exposure. This difference, however, was not statistically significant. See Chapter 11 for further discussion of SO_2 deposition.

13.2.3.4 Subject Activity Level--One practical implication of the above findings is that vigorous activity, such as heavy exercise or work, may significantly affect the actual dose received by a person during exposure to SO_2. Several studies indicate that at some level of ventilation inhalation of air shifts from nasal to oronasal breathing and, also, that some individuals may always be oronasal breathers, even at rest. Saibene et al. (1978), for example, studied 63 subjects while they exercised at increasing workloads. Incomplete information was obtained on 13 subjects. Ten subjects were observed to breath through the mouth at all workloads, while five never opened their mouths. In the remaining 35 subjects, the highest minute volume attained with nasal breathing was 40.2 liters per minute. Determination of the shift from nasal to oronasal breathing was obtained by subjective observation. In a second study using 10 subjects, ventilation was more precisely measured (but still not by a completely adequate technique), as determined by movements of the rib cage. The mean value of ventilation at the point of shift to oronasal breathing was 44.2 liters/min.

Niinimaa et al. (1980, 1981) subsequently reported data indicating that: (1) some subjects are essentially oral breathers even at rest; (2) a switch from nasal to oronasal breathing occurs at Ve of 35.3 ± 10.8 (mean ± standard deviation) liters per minute; and (3) after the switch to oronasal breathing by persistent nasal breathers (at rest), the nasal portion of Ve decreased to 57 percent of total Ve. With further increases in Ve, oral minute volume increased rapidly. Other studies (D'Alfonso, 1980) suggest that subjects who are nasal breathers at rest move to oronasal breathing when minute volume is approximately 30 liters. It should be noted that some air may continue to enter the lungs through the nose after the shift to oronasal breathing, but the nasal volume is substantially reduced (Niinimaa et al., 1980, 1981; D'Alfonso, 1980). Niinimaa et al., for example, reported that as much as 40 to 50 percent of inhaled air entered via the nose after the shift to oronasal breathing. The full implications of these studies on oronasal breathing for understanding the distribution and effects of inhaled pollutants under various activity conditions await further investigation, but they appear to establish that some individuals utilize oronasal breathing even at rest and wide variability exists in regard to Ve levels at which nasal breathers at rest shift over to predominantly oral breathing under increased exercise conditions (i.e., across the range of Ve ~ 15 to 45 liters/minute).

In regard to the impact of increased activity levels on SO_2 effects, Kreisman et al. (1976) reported that exercise may potentiate the effect of SO_2 on respiratory function. In their study, subjects inhaled a mixture of SO_2 in air for 3 minutes while exercising on a bicycle ergometer at a pace sufficient to double their resting minute ventilation rate. Eight subjects received 1 ppm (2.6 mg/m^3) SO_2 and nine subjects received 3 ppm (7.9 mg/m^3). Those receiving 3 ppm (7.9 mg/m^3) showed a significant ($p < 0.05$) decrease in maximal expiratory flow ($MEF_{40\%\ (P)}$) compared to a control (untreated air) exposure. However, it is not clear that this change differed significantly from the change in $MEF_{40\%\ (P)}$ occurring in resting subjects. Bates and Hazucha (1973) reported approximately a 20 percent decrease in maximal expiratory flow rate ($MEFR_{50\%}$) and approximately an 8 to 10 percent decrease in other pulmonary function measurements (FVC, $FEV_{1.0}$, MMFR) in 4 subjects (who exercised intermittently during the exposure) exposed in a chamber containing 0.75 ppm SO_2; however, these differences were not statistically significant. Unlike these studies at 0.75 ppm (2.0 mg/m^3) SO_2, subsequent chamber studies by Hazucha and Bates (1975) at 0.37 ppm (0.97 mg/m^3) SO_2 demonstrated no decrements in any of the above pulmonary function parameters. Horvath and Folinsbee (1977) and Bedi et al. (1979) exposed nine intermittently exercising subjects in a chamber to 0.4 ppm (1.0 mg/m^3) SO_2 and found no pulmonary function changes. Also, more recent studies of exercising adults by Horvath et al. (1981) found neither significant pulmonary function effects with 0.4 ppm SO_2 alone nor any enhancement of ozone effects by 0.4 ppm (1.0 mg/m^3)

SO_2. Analogously, Linn et al. (1980) reported no pulmonary function changes in 24 exercising healthy adults with exposure to 0.5 ppm (1.3 mg/m^3) SO_2 plus 0.5 ppm (0.9 mg/m^3) NO_2 in comparison to exercising baseline control values obtained in an open chamber.

In a recent study by Stacy et al. (1981), sixteen young healthy males were exposed in a chamber for 2 hours to 0.75 ppm (2.0 mg/m^3) SO_2. During the last 15 minutes of the first hour of SO_2 exposure, they exercised at a rate sufficient to increase their ventilatory volumes to 60 liters/minute and SO_2 effects on pulmonary function at the end of the first hour were compared against nonexercise SO_2 exposure baseline test resuts. Airway resistance was significantly increased (mean = 14.6%; range = 2 to 55%) following the exercise period in 14 of the 16 subjects. Although this was the only statistically significant change among the 15 pulmonary measurements made, there was also a trend toward decreased $FEF_{50\%}$ and FEV_1/FVC levels. Half of the SO_2-exposed subjects with one or more positive allergen skin tests appeared to be significantly more reactive to SO_2. Fifteen other subjects were utilized as controls (i.e., being exposed to filtered air and having their data compared to that obtained on the SO_2-exposed subjects). Subjects reacting positively to seven or more skin allergen or metacholine tests were excluded from the study. The statistically significant changes in R_{aw} and notable trends observed by Stacy et al. (1981) in other pulmonary measures appear to imply a bronchoconstriction response occuring in healthy adults (especially atopic subjects without asthma) at 0.75 ppm SO_2 exposure level under heavy exercise conditions.

Lawther et al. (1975) have demonstrated that simply instructing 12 subjects to take 25 deep breaths by mouth resulted in a statistically significant ($p < 0.001$) increase in specific airway resistance (SR_{aw}) during exposure to air alone (14%) and an overall increase of 26% for SO_2 at 1 ppm (2.6 mg/m^3). While sitting quietly in an inhalation chamber, the same subjects had previously shown no such increase after breathing concentrations of 1 to 3 ppm (2.6 to 7.9 mg/m^3) SO_2 for an hour. As part of a series of experiments in this study, 17 subjects also received 3 ppm (7.9 mg/m^3) SO_2 by a mouthpiece and were instructed to take 2, 4, 8, 16 and 32 deep breaths at 5-minute intervals. Increases in SR_{aw} due to SO_2 were significantly greater after 16 ($p < 0.01$) or 32 ($p < 0.001$) deep breaths.

Burton et al. (1969), however, found no consistent effects in 10 subjects exposed to SO_2 at 1.1 to 3.6 ppm (2.9 to 9.4 mg/m^3) for 30 minutes, regardless of whether the subjects breathed normally or at a forced hyperventilation rate of up to 2.5 liters/second. One other difference between these two studies was the duration of exposure. Burton et al. (1969) exposed their subjects for 30 minutes, whereas, Lawther et al. (1975) maintained exposures for an hour. This raises another important consideration in reviewing the effects of SO_2 on human subjects, namely, temporal parameters.

13.2.3.5 Temporal Parameters--Early studies (e.g., Lehman, 1893) suggested that workers chronically exposed to relatively high concentrations of SO_2 were less conscious of its presence in the atmosphere than persons not as familiar with the gas. However, the data of Holmes et al. (1915) indicated that subjects already accustomed to SO_2 could detect its odor at lower concentrations than could persons unaccustomed to it. Nevertheless, it would seem plausible that "self-selection" would tend to reduce the number of relatively sensitive persons among the population of workers chronically in contact with supra-threshold SO_2 levels.

As previously noted, a study by Frank et al. (1962) indicated that subjective reports are not a reliable indicator of physiological responses in any event. After 5 to 10 minutes of exposure to either 5 or 13 ppm (13.1 or 34.1 mg/m^3) SO_2, changes in their subjects' pulmonary resistance measures were just reaching their peaks, while subjective reports of an odor of SO_2 had already subsided.

In a later study by Frank et al. (1964) the increase in pulmonary resistance induced by SO_2 peaked at about 10 minutes and then gradually decreased over the next 15 minutes. This finding corresponds closely to Sim and Pattle's (1957) report that, if lung resistance increased at all in individual subjects, the increase occurred within the first 10 minutes.

Abe (1967) also investigated the temporal course of induction of SO_2-exposure effects. His five mouth-breathing subjects were given 4 to 5 ppm (11 to 13.1 mg/m^3) SO_2; immediate significant ($p < 0.05$) increases in expiratory resistance (42 percent) and inspiratory resistance (25 percent) were observed.

Similar short-term responses (within 5 to 10 minutes after the start of exposure) have been recorded by other investigators. Melville (1970) found in 49 mouth-breathing subjects that percentage decreases in specific airway conductance (SG_{aw}) were greatest during the first 5 minutes of up to 60 minutes of exposure to SO_2. At 5 ppm (13.1 mg/m^3), for example, he noted that SG_{aw} decreased significantly ($p < 0.05$) within 5 minutes of exposure but then stabilized at slightly higher levels that, however, remained significantly lower ($p < 0.05$) during the rest of the one-hour exposure than levels recorded under control conditions of no SO_2.

Lawther et al. (1975) also noted that SR_{aw} increased most during the first 5 minutes of exposure. However, if exposure was terminated after another 5 minutes, recovery to baseline levels generally occurred in about 5 minutes, although 3 "SO_2-sensitive" subjects out of a total of 14 took 10 to 65 minutes to recover from higher exposure levels (up to 30 ppm [78.6 mg/m^3] SO_2) after returning to a clean air environment.

Longer term effects (over a period of hours) have been reported by Andersen et al. (1974), who investigated nasal mucus flow rates as well as airway resistance and subjective responses. Nasal mucociliary flow was measured by placing a radioactive-labeled resin particle on the superior surface of the inferior turbinate and tracking its position with a slit-collimator detector. Fifteen subjects were exposed via an inhalation chamber to increasing concentrations (1, 5, and 25 ppm [2.6, 13.1 and 65.5 mg/m^3]) of SO_2 for approximately 6 hours per day over 3 consecutive days. Baseline measurements were made under conditions of filtered air on a day prior to experimental exposures. This study found a number of effects reaching their maximum after 1 to 6 hours of exposure. Nasal cross-sectional airway area generally decreased throughout the 6-hour daily trials, but the decreases were only significant ($p < 0.05$)

at 1 ppm (2.6 mg/m^3) and 5 ppm (13.1 mg/m^3), since there was an overall drop in this measure (approaching a "floor level") by the time 25 ppm (65.5 mg/m^3) was administered on the third day of the study. Nasal airflow resistance increased significantly ($p < 0.05$) with the 6-hour exposure to each concentration (1, 5, 25, ppm [2.6, 13.1, 65.5 mg/m^3] SO_2). Significant ($p < 0.05$ or less) decreases in forced expiratory flow ($FEF_{25-75\%}$) and forced expiratory volume ($FEV_{1.0}$) also occurred both within daily exposures and across days (i.e., increasing concentrations), although the within-day decrease in $FEV_{1.0}$ was only significant on day 3 at 25 ppm [65.5 mg/m^3] (see Andersen et al., 1974, Figure 7).

13.2.3.6 Mucociliary Transport--Cralley (1942) investigated mucociliary clearance before sophisticated radioactive measurement techniques were available. A drop of red dye was placed in the active ciliary region of the inferior meatus of a volunteer subject. The rate of mucus clearance was reflected by the time between introduction of the dye and its appearance in expelled mucus. Exposure to SO_2 at 10 to 15 ppm (26.2 to 39.3 mg/m^3) for 60 minutes produced only a small decrease in the rate of mucus removal. A 30- to 60-minute exposure resulted in a 50 percent reduction in mucociliary transport at 25 ppm (65.5 mg/m^3) SO_2 and a 65 to 70 percent reduction at 50 ppm (131 mg/m^3).

In another study (Anderson et al., 1974), mucostasis in the anterior region of the nose was observed in 14 of 15 subjects after 4 to 5 hours of exposure on three successive days to 1, 5, and 25 ppm (2.6, 13.1 and 65.5 mg/m^3) SO_2, respectively. There appeared to be no carryover effect from the exposure of the previous day. Mucus flow rates on the first day of exposure (to 1 ppm [2.6 mg/m^3] SO_2) tended to be lower but were not significantly lower than those observed on the control day (0 ppm). Mucus flow rates were significantly lower on the second day (5 ppm [13.1 mg/m^3]) and were further decreased on the third day (25 ppm [65.5 mg/m^3]) of exposure. The subjects noted discomfort only on the second and third day exposures. At these concentrations some subjects also had sporadic mucostasis, although there were pronounced individual differences in these measures even at baseline. Andersen et al. (1974) calculated the cross-sectional area of the nasal airways. A statistically significant decrease ($0.02 < P < 0.05$) was observed during the 1 ppm exposure. The reduction in area during the 4 to 6 hour exposure to SO_2 amounted to 11, 24, and 29 percent at 1, 5, and 25 ppm (2.6, 13.1 and 65.5 mg/m^3) SO_2 respectively.

Wolff and his co-workers (Wolff et al., 1975a,b; Newhouse et al., 1978) have also measured the rate of mucociliary transport. In the first study by Wolff et al. (1975a), nine subjects were exposed to 5 ppm (13.1 mg/m^3) SO_2 for 2 hour while sitting quietly in an inhalation chamber and breathing through their mouths. Mucociliary clearance was assessed by having the subjects first inhale a radioactively tagged aerosol and then monitoring its subsequent tracheobronchial retention during SO_2 exposure. No significant effects were found on mucociliary clearance, except for a small transient change ($p < 0.05$) after 1 hour of exposure.

In their second study, Wolff et al. (1975b) used similar methods to compare subjects while resting or exercising. Exercise was performed on a bicycle ergometer for 0.5 hour at a pace to yield heart rates 70 to 75 percent of estimated maximum values. Exposure in this study lasted for 2.0 to 2.5 hours. The combination of exercise and exposure (via mouth) to 5 ppm (13.1 mg/m^3) SO_2 resulted in a significantly ($p < 0.05$) greater rate of tracheobronchial mucociliary clearance. This result contrasts with the findings of Andersen et al. (1974) that nasal clearance rates were reduced by exposure to 5 ppm (13.1 mg/m^3) SO_2. However, the difference between the two studies can probably be explained on the basis of dose. Dose to the lung would be much lower than to the nose because of the absorption of SO_2 by upper airway mucosal surfaces. Therefore, effects on the lung could be typical of lower concentrations and increases might be anticipated as seen for low levels of H_2SO_4. Of course, the two studies focused on different regions of the respiratory tract (tracheobronchial versus nasal), and this in itself could account for these contrasting effects (Albert et al., 1969; Frances et al, 1970). Both of these investigators replicated their findings in later studies (Andersen et al., 1977; Newhouse et al., 1978).

These studies were extended by Newhouse et al. (1978), whose 10 subjects breathed either SO_2 (5 ppm) or H_2SO_4 mist (1 mg/m^3) delivered as an aerosol of 0.58 µm MMAD. An aerosol containing a 0.025 percent solution of ^{99m}Tc-albumen was inhaled prior to pollutant exposure. The bolus technique (exposure to short-term peak concentrations) employed achieved deposition of the aerosol, primarily in the large airways. One-half hour later the subjects were exposed to the pollutants. They immediately exercised for the next 0.5 hour. A total of 20 minutes of exercise at approximately 70 to 75 percent of predicted maximum heart rate was performed, followed by an additional 1.5 hours of rest exposure. The subjects breathed through the mouth to eliminate nasal ventilation and absorption of pollutants. Pulmonary function tests conducted at the end of 2 hours exposure to SO_2 indicated no changes in FVC or $FEV_{1.0}$ but maximum midexpiratory flow rate (MMFR) decreased 8.5 percent, possibly due to a reflex bronchoconstriction. No pulmonary changes were found consequent to the H_2SO_4 mist exposures. Tracheobronchial clearance increased in both SO_2 (6 of 10 subjects) and H_2SO_4 (5 of 10 subjects) exposures. The investigators did not present their data in a manner which would provide information as to the relationship between clearance rates and MMFR. It should be noted that these data are in contrast to the replicated observations by Andersen et al. (1977), who showed a slowing of nasal clearance on exposure to 5 ppm (13.1 mg/m^3) SO_2. It is possible that time-dependent events (or subject selection) might explain the different results; or differences in the respiratory tract regions studied (tracheobronchial versus nasal) may account for the apparent inconsistency.

Mucociliary transport is a significant aspect of the respiratory system's defense against airborne agents. A disturbance in this function might have important implications for a number of health effects, such as susceptibility to cold-virus infections. Andersen et al. (1977) evaluated this possibility by inoculating volunteers with a strain of rhinovirus (RV3). The basic design of the study and reactions of the subjects are shown in a table (Andersen et al., 1977). Although there was no difference in the number of colds that developed in the two groups of subjects (all nose breathers), cold symptoms were judged (under a double-blind procedure) to be less severe ($p < 0.05$) in the group exposed to SO_2. It was unknown, however, whether this result reflected a direct effect of SO_2 on the host, the rhinovirus, or both. In addition, the average incubation period was somewhat shorter for the group exposed to SO_2 ($p < 0.06$). Virus shedding (a measure of infection determined from nasal washings) also seemed to be somewhat decreased in the SO_2-exposed group but not significantly.

SO_2 may affect mucociliary clearance via direct effects on secretory or ciliated cells or indirectly through SO_2 effects on neural reflexes involved in clearance control. Related to the latter, Camner et al. (1974, 1976) evaluated the role of the autonomic nervous system in the regulation of tracheobronchial clearance in man. They employed a test aerosol of 6 μm Teflon particles tagged with technetium 99m. Their first study (1974) indicated the basic mucociliary transport was not under parasympathetic control. However, activation of the vagus did induce increases in transport velocity. Their second study (1976) suggested that the regulation of the mucociliary transport rate was under sympathetic control and blood concentrations of catecholamines might be important in modulation of mucociliary transport. This is in contrast to the likelihood that parasympathetic cholinergic neural mechanisms are of key importance in mediation of SO_2-induced bronchoconstriction effects.

13.2.3.7 Health Status--Apart from fairly consistent bronchoconstriction effects, a common element in the above and other reports of the effects of SO_2 has been the notable variability among subjects in their responses to such exposures. In the study of Frank et al. (1962), for example, 9 of 11 subjects showed no effects at 1 ppm, but 1 subject showed a significant ($p < 0.01$) decrease in pulmonary flow resistance, whereas the remaining subject showed a significant ($p < 0.01$) increase. Sim and Pattle (1957) reported that they themselves appeared to have developed an increasing sensitivity to SO_2 encountered during the course of their research, especially upon initial contact with the gas. The persistent and uncomfortable spells of coughing and wheezing upon contact with the gas appeared to increase with repeated exposures. Other investigators (e.g., Burton et al., 1969; Frank, 1964; Nadel et al, 1965; Lawther, 1955; Lawther et al., 1975; Jaeger et al., 1979) have reported "hyperreactors" among their subjects. Indeed, some investigators have suggested that about 10 percent of the total population is made up of especially sensitive persons (Amdur, 1973, 1974; Horvath and Folinsbee, 1977). However, in at least one instance (Andersen et al., 1974), a subject's response was exaggerated even under control conditions, which raises the possibility of psychological factors contributing to this observed sensitivity.

Reports of the prevalence of asthma within the population vary with the criteria used to identify asthmatics, differences in study populations, and other factors. Data from the 1970 Health Interview Survey indicate the prevalence of asthma to be about 3 percent of the population. However, Dodge and Burrows (1980) reported that, in a sample of white (non-Mexican) individuals living in Tucson, Arizona, some 6.6 percent had physician-diagnosed asthma and another 9 percent had spontaneous attacks of wheezing and shortness of breath. Another 10 percent had wheezing without colds, and 10 percent had wheezing with colds. The generally cited figure of 8-15 percent of the population having asthma may, therefore, be too low. It should be recalled that most investigators of cinical studies have reported that 10 to 20 percent of their "normal" subjects have shown unusual sensitivity to SO_2 exposure.

Some studies have considered the preexisting health status of subjects as a variable in assessing the physiological effects of SO_2. Weir and Bromberg, for example, conducted separate studies on 12 healthy subjects (Weir and Bromberg, 1972) and on 7 smokers who showed early signs of chronic obstructive pulmonary disease (Weir and Bromberg, 1973). The subjects were exposed to 0, 0.3, 1, and 3 ppm (0, 0.8, 2.6 and 7.86 mg/m^3) SO_2 in an inhalation chamber for 96 or 120 hours (smokers or nonsmokers, respectively), with several days separating each trial. In non-smoking, healthy subjects there was a significant, but reversible effect on the pulmonary function of adult males at 3 ppm SO_2 exposure for 120 hours. The individual variablity among the smokers in their daily lung functions was so great that no effects could be attributed to SO_2 exposure. Also, subjective complaints appeared to be randomly distributed throughout the course of the study and could not be related to SO_2 exposure levels. Gökenmeyer et al. (1973), however, reported that bronchitic patients exposed to 10 ppm (26.2 mg/m^3) SO_2 had maximal changes in SR_{aw} at the end of a 3-minute period of inhalation. Recovery to control levels required some 45 to 60 minutes when subjects were returned to a clean-air environment.

Reichel (1972) exposed two groups of subjects with obstructive bronchial disease to varying concentrations of SO_2 in his chamber. Patients with minor obstructive disease were exposed continuously for 4 days to 3.81 ppm SO_2 (10 mg/m^3, n= 8), for 4 days to 1.80 ppm SO_2 (4.7 mg/m^3, n=4) and for 6 days to 0.29 ppm SO_2 (0.75 mg/m^3, n = 5). Patients with serious obstructive bronchial disease were exposed to 1.52 to 1.80 ppm SO_2 (4 to 4.7 mg/m^3) for 4 days and 1 to 1.52 ppm SO_2 (2.6 to 4.0 mg/m^3) for 6 days. Airway resistance was not influenced by such exposure. The details of the measuring procedures were not adequately presented in his report.

Gunnison and Palmes (1974) compared 7 heavy smokers and 13 non-smokers with respect to blood plasma levels of S-sulfonate after exposure to 0.3, 1.0, 3.0, 4.2, and 6.0 ppm (0.8, 2.6, 7.9, 11.0, and 15.7 mg/m^3) SO_2. Both groups showed highly significant correlations ($p < 0.001$) between SO_2 concentrations and S-sulfonate levels. But there was no significant differentiation between the two groups of subjects in this regard.

Jaeger et al. (1979) exposed 40 asthmatics (mild to moderate but with no recent exacerbations) for 4 hours to 0.5 ppm (1.3 mg/m^3) SO_2. Oral inhalation was forced by having the resting subjects wear nose clips. Control studies were made during exposure to ambient air having an average SO_2 of 0.005 ppm (13.1 µg/m^3). The only statistically significant effect ($p < 0.04$) observed was a 2.7-percent decrease in MMFR after 0.5 ppm (1.3 mg/m^3) SO_2 exposure.

This minimal change was stated to have little physiological significance. Two asthmatic subjects exhibited discomfort and audible wheezing (requiring standard asthmatic medication) during the night following SO_2 exposure; again, however, attribution of these "delayed" symptoms to the earlier SO_2 exposure may be questionable.

Sheppard et al. (1980) exposed three groups of seven subjects (normal, atopic, and mild asthmatic) for 10 minutes to 0, 1, 3, and 5 ppm (0, 2.6, 7.9 and 26.2 mg/m^3) SO_2. The subjects breathed these gases orally, via mouthpieces, while their specific airway resistance (SR_{aw}) was measured in a body plethysmograph. The intermittent exposures to SO_2 consequent to the methods used may have influenced the results. Several statistical approaches were utilized, one of which may be inappropriate, but the general conclusions appear to be valid. Despite large inter- and intrasubject variability in these subjects breathing clean air, it was found that in asthmatic subjects SR_{aw} increased significantly ($p < 0.05$ to 0.025) at all concentrations of SO_2. Normal and atopic (skin sensitive to common allergens) subjects had statistically significant increases in SR_{aw} only while breathing 5 ppm (13.1 mg/m^3) SO_2. Some asthmatic subjects exhibited marked dyspnea requiring bronchodilator therapy. The increased SR_{aw} seen in either normal or mild asthmatic subjects was prevented by treatment with atropine, confirming the involvement of parasympathetic pathways in this response.

Sheppard et al. (1981), using 13 non-smoking mildly asthmatic volunteers (10 men, 3 women, 20 to 30 years of age), demonstrated that moderate exercise (minute ventilation ~ 30 liters) increased the bronchoconstriction effect of SO_2 administered by mouth at SO_2 concentrations of $\leq$ 1 ppm (2.6 mg/m^3). The first set of studies evaluated the effect of exercise on SO_2-induced bronchoconstriction in seven subjects (six men, one woman) at concentrations of 0.5, 0.25, and 0.1 ppm (1310, 660 and 260 μg/m^3). In these subjects, inhalation of 0.50 and 0.25 ppm (1310 and 660 μg/m^3) of SO_2 during the performance of moderate exercise significantly increased SRaw, whereas neither inhalation of 0.50 ppm (1.3 mg/m^3) of SO_2 at rest nor inhalation of humidified, filtered air during exercise had any effect on SRaw. Inhalation of 0.50 ppm (1.3 mg/m^3) during exercise significantly increased SRaw in all seven asthmatic subjects ($p < 0.05$), and three developed wheezing and shortness of breath. During the corresponding period of exercise alone and during inhalation of 0.50 ppm (1.3 mg/m^3) at rest, SRaw did not increase in any subject. After inhalation of 0.50 ppm (1.3 mg/m^3) of SO_2 during exercise, SRaw was significantly greater than after exercise alone or inhalation of 0.50 ppm of SO_2 at rest ($p < 0.05$). Inhalation of 0.25 ppm (660 μg/m^3) during exercise significantly increased SRaw in three of the seven subjects, and the increase in SRaw for the group was significant ($p < 0.05$). No subject developed wheezing or shortness of breath at 0.25 ppm (660 μg/m^3) SO_2 exposure. During the corresponding period of exercise alone, SRaw did not increase in any subject. In the two most responsive subjects at 0.5 (1310 μg/m^3) and 0.25 (660 μg/m^3) SO_2, inhalation of 0.1 ppm (260 μg/m^3) SO_2 significantly increased SRaw, and there appeared to be a dose-response relationship of successively greater increases in SR_{aw} as a function of increasing SO_2 concentrations (0.1, 0.25, and 0.50 ppm).

The second set of studies, involving all six subjects (four men, two women), compared the bronchoconstriction produced by breathing SO_2 (1 ppm [2.6 mg/m^3)] during exercise and during eucapnic hyperventilation. The magnitude of the increase in SR_{aw} was the same when the subjects inhaled SO_2 while they exercised or when they performed eucapnic hyperventilation at the same minute ventilation. In every case, the increase in SRaw was accompanied by dyspnea and audible wheezing.

The bronchoconstriction produced by inhalation of 0.50 ppm (1.31 mg/m^3) SO_2 during exercise was gradual in onset. Immediately after exercise, SRaw did not differ significantly from baseline values. It then increased over the first 3.5 minutes, reached a plateau, and gradually returned to baseline values by 30 minutes after exposure. A similar time course was seen in those subjects who developed bronchoconstriction after exposure to 0.25 and 0.10 ppm (660 and 260 $\mu g/m^3$) SO_2. In contrast, the bronchoconstriction produced by inhalation during eucapnic hyperventilation was rapid in onset, suggesting that the bronchoconstriction observed during exercise was a function of the increases in minute ventilation. These investigators utilized two different statistical procedures to analyze their data. It is not clear which was utilized for each of their conclusions.

Linn et al. (1982) confirmed qualitatively the results of Sheppard et al. (1980). Five asthmatic subjects were exposed to 0.5 ppm (1.3 mg/m^3) SO_2 via mouthpiece for periods of 10 minutes during moderate exercise (~ 400 kg-m/min.). Compared to baseline conditions employing clean air plus exercise, all subjects except one showed greater increases in specific airway resistance while breathing 0.5 ppm SO_2 than while breathing clean air. Also in a larger scale followup study employing open chamber exposures, Linn et al. (1982) measured pulmonary functions (airway resistance, forced vital capacity) and recorded various symptoms (cough, sputum production, wheezing, chest tightness, substernal irritation, dyspnea, throat irritation or congestion, headache, eye irritation, and fatigue) in 24 young adult asthmatic subjects exposed to sulfur dioxide (0.25 ppm and 0.5 ppm) for 1 hour under intermittent periods (10 minutes) of moderate exercise (Ve ~ 27 liters/minute). None of the measurements of pulmonary function (SR_{aw} and FVC) showed statistically significant variation attributable to sulfur dioxide, although small significant increases in resistance attributable to exercise were found. Similar results were observed by Linn et al. (1980) in earlier chamber studies employing 19 asthmatics exercising at workload levels sufficient to approximately double their resting minute ventilation rates (normal Ve $\simeq$ 8 to 10 liters/minute). No changes in pulmonary functions were observed over exercise baseline rates when the subjects were exposed to 0.3 ppm (0.79 mg/m^3) SO_2 in combination wtih 0.5 ppm (0.9 mg/m^3) NO_2 while exercising in an open exposure chamber.

The Linn et al. (1980, 1982) chamber study results are in contrast to results obtained in the mouthpiece exposure studies conducted by Sheppard et al. (1981) and Linn et al. (1982). These differences are most likely accounted for by greater doses of SO_2 reaching tracheobronchial regions with mouthpiece exposure than with open chamber exposures at slightly lower exercise levels. Also, individual variation in bronchial reactivity to SO_2 among the subjects tested may have contributed to the contrasting results.

Several other studies of SO_2 (e.g., Snell and Luchsinger, 1969; Andersen et al., 1974; Gokenmeijer et al., 1973; Burton et al., 1968, 1969) have included asthmatic patients or smokers but have not provided even qualitative ratings of their health status. This alone makes it difficult to compare the results of these to other studies using "healthy" or "impaired" subjects. Morever, the great individual variability among both normal and impaired persons in these particular studies makes it difficult to reach any conclusions based on their results regarding the relative importance of an individual's health status in determining his physiological response to SO_2.

13.3. PARTICULATE MATTER

As chemical interactants, particles such as aerosols of certain soluble salts (e.g., ferrous iron, manganese, vanadium) may act as catalysts to convert SO_2 to H_2SO_4. Water from atmospheric humidity or from physiological sources figures prominently in these reactions. The following sections deal with common compounds of sulfur dioxides and point up the influence of a number of variables that affect human physiological response to these compounds.

13.3.1 Sulfuric Acid and Sulfates

13.3.1.1 Sensory Effects--A number of studies have been directed toward determining threshold concentrations of H_2SO_4 for various sensory response. In a study with 10 test subjects, Bushtueva (1957) found that the minimum concentration of sulfuric acid aerosol (particle size not given) which was sensed by odor ranged from 0.6 mg/m^3 to 0.85 mg/m^3 (average 0.75 mg/m^3). In tests with five subjects (Bushtueva, 1961), a combination of sulfur dioxide at 1 mg/m^3 (0.35 ppm) and sulfuric acid mist at 0.4 mg/m^3 was below the odor threshold. Amdur et al. (1952) reported on 15 subjects (males and females) exposed for 5 to 15 minutes to various concentrations of sulfuric acid mist that the subjects breathed via face masks. It was found that 1 mg/m^3 was usually not detected, while 3 mg/m^3 was detected by all subjects.

Bushtueva (1957) studied the effect of sulfuric acid mist on the light sensitivity of two test subjects. Sensitivity was measured every 5 minutes during the first half-hour of each test, then at 10-minute intervals thereafter. A control curve was established for each subject by seven repeated tests, and then sulfuric acid aerosol was administered for 4 minutes and for 9 minutes at the 15th and 60th minutes, respectively. With sulfuric acid mist of undetermined particle size at a concentration of 0.6 mg/m^3, a just detectable increase in light sensitivity occurred with the first exposure but not with the second. Concentrations in the range of 0.7 mg/m^3 to 0.96 mg/m^3 brought about a well-defined increase in light sensitivity. With 2.4 mg/m^3, increased sensitivity to light was elicited by the exposures at both the 15th and 60th minutes of the test; normal sensitivity was restored in 40 to 50 minutes.

Bushtueva (1961) studied the effect of sulfur dioxide, sulfuric acid mist and combinations of the two on sensitivity of the eye to light in three subjects. The combination of sulfur dioxide at 0.65 mg/m^3 (0.23 ppm) with sulfuric acid mist at 0.3 mg/m^3 resulted in no change in sensitivity of the eye to light. An increase of approximately 25 percent in light sensitivity resulted from exposure to either sulfur dioxide at 3 mg/m^3 (~1.0 ppm) or sulfuric acid mist at 0.7 mg/m^3. The combination of sulfur dioxide at 3 mg/m^3 (~1.0 ppm) with sulfuric acid mist at 0.7 mg/m^3 resulted in an increase of approximately 60 percent in light sensitivity. Exposures lasted for 4½ minutes.

Bushtueva (1962) demonstrated that combinations of sulfur dioxide at 0.50 mg/m^3 (0.17 ppm) with sulfuric acid mist at 0.15 mg/m^3 or sulfur dioxide at 0.25 mg/m^3 (0.087 ppm) with sulfuric acid mist at 0.30 mg/m^3 could produce electrocortical conditioned reflexes. There are some uncertainties regarding this study.

Bushtueva (1961) studied the effects of different concentrations of sulfur dioxide, sulfuric acid mist, and combinations of the two on the optical chronaxie of three subjects. Optical chronaxie was determined in each test subject at 3-minute intervals as follows: at the start and on the 3rd, 6th, 9th, 12th and 15th minutes. Between the 6th and 9th minutes the subjects inhaled sulfur dioxide, sulfuric acid mist, or their combination for 2 minutes. In each subject, the threshold concentrations of sulfur dioxide and sulfuric acid mist were first determined independently, and then threshold concentrations for combinations of the two were determined. Sulfuric acid mist (0.75 mg/m^3) increased optical chronaxie.

13.3.1.2 Respiratory and Related Effects--Studies investigating respiratory and related effects of human exposures to sulfuric acid under controlled conditions are summarized in Table 13-3. Amdur et al. (1952) found respiratory changes in all subjects exposed for 15 minutes to H_2SO_4 aerosol at concentrations of 0.35 mg/m^3 to 5 mg/m^3. Vapors from an electrically heated flask containing concentrated sulfuric acid were carried by compressed air into the main air stream and then into a lucite mixing chamber, delivering a mist with a mass median diameter of 1 µm. The subjects breathed through a pneumotachograph, permitting measurement of inspiratory and expiratory flow rate. In 15 subjects, exposed to 0.35, 0.4, or 0.5 mg/m^3, the respiration rate increased about 35 percent above control values, while the maximum inspiratory and expiratory flow rates decreased about 20 percent. Tidal volume decreased about 28 percent in subjects exposed to 0.4 mg/m^3. These changes occurred within the first 3 minutes of exposure and were maintained throughout the 15-minute exposure period. Lung function returned rapidly to baseline levels after the exposure ended. The tidal volume rose above control values during the first minute after termination of the exposure and then returned to preexposure levels. Breathing through the same apparatus without the acid mist was done as a control, and no such changes were observed. Some subjects showed a marked reaction to 5 mg/m^3, a level of acid mist perceptible to all. Individual responses were much more varied at this level, the main effect being a decrease in minute volume. The investigators suggest that bronchoconstriction may have been the response to sulfuric acid.

The effect of breathing sulfuric acid mist at different relative humidities (RH) was studied by Sim and Pattle (1957). Healthy males (variable number of subjects), 18 to 46 years

13-3. PULMONARY EFFECTS OF SULFURIC ACID

Concentration	Duration of exposure (mins)	Number of subjects	Source	Effects	Reference
0.35 - 5.0 mg/m^3 H_2SO_4 MMD 1 µm	15	15	Mask (rest)	Respiratory rates increased, max. insp. and expiratory flow rates and tidal volumes decreased	Amdur et al., 1952
3-39 mg/m^3 H_2SO_4 MMD 1-1.5 µm	10 - 60	Variable	Mask (rest) Chamber (rest)	Longer particles due to "wet mist" resulted in increased flow resistance, cough, rales and bronchoconstriction	Sim and Pattle, 1957
SO_2 (1-60 ppm) plus H_2O_2 to form H_2SO_4 aerosol CMD 1.8 and 4.6 µm	Variable	24	Mask (rest) Oral	Airway resistance increased especially with larger particles	Toyama and Nakamura, 1964
H_2SO_4 mist (1000 µg/m^3 MMD 0.5 µm (σg = 2.59)	120	10	Chamber (exercise) Oral	No pulmonary function changes but increased tracheobronchial clearance	Newhouse et al., 1978
H_2SO_4 aerosol 10, 100, 1000 µg/m^3 MMD 0.1 µm	10	6 normal 6 asthmatics	Oral*	No pulmonary function changes, no alterations in gas transport	Sackner et al., 1978
H_2SO_4 (75 µg/m^3) MMAD 0.48 - 0.81 µm	120	6 normal 6 asthmatics	Chamber (exercise)	No pulmonary effects in either group	Kleinman and Hackney, 1978; Avol et al., 1979
H_2SO_4 (0, 100, 300, or 1,000 µg/m^3 MMAD 0.5 µm (σg = 1.9)	60	10	Nasal (rest)	No pulmonary function effects. Bronchial mucociliary clearance ↑ following 100 µg/m^3 but ↓ following 1000 µg/m^3; mucociliary clearance distal to trachea more affected	Leikauf et al., 1981
H_2SO_4 (0, 223, 418, 939 µg/m^3 MMD 0.90 - 0.93 µm (σg = 1.66 - 1.73)	120	11 (2 exsmokers; 6 allergics; 1 childhood asthmatic; 2 normals)	Chamber (exercise)	Small statistically significant change only in $FEV_{1.0}$ at 939 µg/m^3 but physiological significance questioned.	Horvath et al., 1981

TABLE 13-3. (continued)

Concentration	Duration of exposure (mins)	Number of subjects	Source	Effects	Reference
H_2SO_4 @ 100 μg/m³ MMD 0.14 μm σg = 2.9	240	28 normals	Chamber (exercise)	No pulmonary function effects	Kerr et al., 1981
High conc. aerosol (1 mg/m³ each) Low conc. aerosol (0.1 mg/m³ each) MMAD = 0.5-1.0 μm Aerosols included: $NaHSO_4$ NH_4HSO_4 $(NH_4)_2SO_4$ H_2SO_4	16	16 normals 17 asthmatics	Oral* (rest)	SG_{aw} induced by carbachol sig. potentiated in asthmatics at 1 mg/m³ H_2SO_4 and NH_4HSO_4 each. $FEV_{1.0}$ sig. decreased after H_2SO_4 and NH_4HSO_4. No changes in SG_{aw} with all sulfates; but two most responsive asthmatics to high H_2SO_4 dose exhibited potentiation effect on carbahol-induced bronchoconstriction at lower H_2SO_4 level.	Utell et al., 1981

*Mouthpiece

of age, breathed 3 to 39 mg/m^3 concentrations of H_2SO_4 at 62 percent RH either via mask or exposure chamber. Subjects were also exposed in the chamber to 11.5 to 38 mg/m^3 concentrations at 91 percent RH. At the lower RH, particles were 1 µm in size. The addition of water vapor to raise RH increased the mean particle size to 1.5 µm and intensified irritant effects of exposure. For example, the irritancy of wet mist at 20.8 mg/m^3 was much more severe ("almost intolerable at the onset") than that of the dry mist at 39.4 mg/m^3 ("well tolerated by all"). Air flow resistance ranged from 43 to 150 percent above normal in response to the wet mist, compared to increases ranging from 35.5 to 100 percent above normal in response to the dry mist. Two subjects exposed to sulfuric acid mist developed bronchitic symptoms but may have been previously exposed to other substances. Adding ammonia (quantity not given) to the acid mist annulled its irritant properties. There was no consistent evidence that the acid mist caused changes in respiratory functions or blood pressure, pulse rate, or other cardiovascular functions.

Toyama and Nakamura (1964) investigated the synergistic effects of SO_2 in combination with hydrogen peroxide (H_2O_2) aerosol mixtures, the latter of which oxidizes SO_2 to form SO_3, which reacts with moisture (H_2O) to form H_2SO_4. Sulfur dioxide concentrations ranged from 1 to 60 ppm (2.6 to 157 mg/m^3); the H_2O_2 concentrations were 0.29 mg/m^3 for particles of 4.6 µm CMD (Horvath estimated MMAD was 13) and 0.33 mg/m^3 for particles of 1.8 µm CMD (Horvath estimated MMAD was 5). Airway resistance increased significantly in the combination (H_2O_2 + SO_2) exposure, particularly for the group of 15 subjects inhaling the larger particles ($p < 0.01$). Toyama and Nakamura (1964) exposed subjects to a mixture of SO_2 and H_2SO_4 aerosols. They used an inadequate method to measure airway resistance. They described the aerosols as having a 4.5 µm diameter. They found a strong constricting effect on the upper airways.

Sackner et al. (1978) studied normal resting young adults and seven asthmatic middle-aged subjects who breathed, by mouth, either sodium chloride or sulfuric acid aerosols for 10 minutes at concentrations of 10, 100, and 1000 µg/m^3 (0.1-0.2 MMAD). Measurements on these individuals continued for up to 3 hours after exposure. The asthmatic patients represented a wide range of clinical status and treatment. Neither normal nor asthmatic individuals showed statistically significant alterations of lung volumes, distribution of ventilation, earoximetry, dynamic mechanics of breathing, oscillation mechanics of the chest-lung system, pulmonary capillary blood flow, diffusing capacity, arterial oxygen saturation, oxygen uptake, or pulmonary tissue volume. No delayed effects were observed during a follow-up period of a few weeks.

Kleinman and Hackney (1978) and Avol et al. (1979) reported on the pulmonary responses of six normal subjects exposed in an ambient environment of 88°F dry bulb and 40 percent relative humidity, and 94 µg/m^3 H_2SO_4. A sham exposure was followed by 2 consecutive days of acid exposure. Sufficient excess acid aerosol to neutralize the NH_3 present (about 56 µg/m^3 ammonia neutralization product) was added to the air to provide for the desired acid concentration (75 µg/m^3). The aerosol MMAD was approximately 0.48 to 0.81 µm. The effective exposure time was

2 hours, with the first 15 minutes of each half hour devoted to exercise that increased ventilation to twice the resting level. Only one subject was exposed at a time to minimize the effects of ammonia neutralization. The normal subjects showed no exposure-related changes.

Utell et al. (1981) exposed 16 normal subjects (all subjects non-smokers) to acidic aerosols, each at a concentration of 1 mg/m^3, (MMAD = 0.5-1.0 μm, σ_g 1.5-2.2) for periods of 16 minutes. Several aerosol exposures were given each day in a double-blind random pattern. At the beginning of each study, an approximate dose-response curve to inhaled carbachol was obtained. All aerosols [$NaHSO_4$; $(NH_4)_2SO_4$; NH_4HSO_4; H_2SO_4; NaCl] were given orally. Although normals demonstrated no reduction in airway conductance, the bronchoconstrictor action of carbachol was significantly potentiated by the prior inhalation of sulfuric acid or ammonium bisulfate. The bronchioconstriction action of carbachol was potentiated by the sulfate aerosols in proportion to their acidity.

Lippmann and coworkers (Lippman et al., 1980; Leikauf et al., 1981) had 10 non-smokers inhale via nasal mask 0.5 μm (σ_g = 1.9) H_2SO_4 at 0 and approximately 100, 300, and 1000 μg/m^3 for 1 hour. The exposures were random over the 4 days of testing. Pulmonary functions (assessed by body plethysmograph, partial forced expiratory maneuver, and nitrogen washout) were measured before exposure, and at 0.5, 2, and 4 hours postexposure. A ^{99m}Tc-tagged monodispersed Fe_2O_3 aerosol (7.5 μm MMAD, σ_g = 1.1) was inhaled 10 minutes before exposure for the determinations of lung retention of these particles. Tracheal mucus transport rates (TMTR) and bronchial mucociliary clearance were determined. No statistically significant changes in respiratory mechanics or TMTR were observed following H_2SO_4 exposure at any level. However, bronchial mucociliary clearance halftime ($TB_{1/2}$) was on the average markedly altered upon inhaling H_2SO_4 at concentrations of 100 and 1000 μg/m^3. Bronchial clearance was accelerated ($p < 0.02$) following exposure to 100 μg/m^3 H_2SO_4, while following exposure to 1000 μg/m^3, it was significantly ($p < 0.03$) retarded. Mucociliary transport in the airways distal to the trachea was affected more by H_2SO_4 exposure than was transport in the trachea. Out of ten subjects, four did not respond. These four had the fastest clearance rates of the ten subjects in their control tests. They were retested at 1000 μg/m^3, with the H_2SO_4 exposure preceding the radiolabelled aerosol exposure, in order to determine if the H_2SO_4 effect had occurred too late in the tagged particle clearance to have affected the measurement. In these followup tests, three of the four responded with a slowing of mucociliary clearance comparable in magnitude to that seen in the other subjects. Thus, there was only one subject in the ten whose clearance was not markedly affected by exposure to H_2SO_4. The alterations in bronchial clearance halftime were all transient, which was consistent with the results seen earlier in similar inhalation tests on donkeys (Schlesinger et al., 1978). However, when donkeys were repeatedly exposed to sulfuric acid at comparable concentrations, four of six animals developed persistently slowed clearance, which remained abnormal for at least several months (Schlesinger et al., 1978, 1979). Taken together, these results suggest that under chronic exposure conditions at the concentrations employed, persistent changes could occur in mucociliary clearance in previously healthy individuals and exacerbate preexisting respiratory disease.

Kleinman and Hackney (1978) and Avol et al. (1979) presented in greater detail the preliminary findings reported by Bell and Hackney (1977a). They evaluated the effects of various sulfate compounds on normal subjects, ozone-sensitive subjects, and asthmatic subjects (requiring medical treatment). The exposures were approximately 2.5 hours in duration, with the subjects exercising the first 15 minutes of each half hour at a pace sufficient to double their ventilation rates. Measurements of pulmonary functions, which included FVC, FEV_1, MEFR, $FEF_{50\%}$, $FEF_{75\%}$, TLC, RV, delta nitrogen (ΔN_2), closing volume, and total respiratory resistance (R_t) were made before and 2 hours after the work-rest regimen began. The ambient conditions were 88°F dry bulb and either 40 or 85 percent relative humidity. Most of the exposure studies were made on five to seven subjects. Four to five sensitive subjects and six asthmatics completed the subject pool. Subjects were first exposed to a control (no pollutant) environment and then to 2 or 3 consecutive days of the pollutants. Nominal exposure concentrations were 100 $\mu g/m^3$ for ammonium bisulfate (NH_4HSO_4) and 85 $\mu g/m^3$ for ammonium sulfate [$(NH_4)_2SO_4$]. The sulfate aerosol size distribution was nominally 0.4 μm MMAD (σ_g 2.5 to 3). There was some ammonia (NH_3) in the exposure chamber. Pulmonary functions were unaffected by exposure to the two types of aerosol.

Kleinman et al. (1981) conducted studies in which 19 volunteers with normal pulmonary function and no history of asthma were exposed on two separate days to clean air and to an atmosphere mixture containing O_3 (0.37 ppm), SO_2 (0.37 ppm), and H_2SO_4 aerosol (100 $\mu g/m^3$, MMAD 0.5 μm; σg = 3.0). Chemical speciation data indicate that 93 percent of the sulfuric acid aerosol had been partially neutralized to ammonium bisulfate. Additional data suggest that the acidity of the aerosol in the chamber decreased as a function of time during exposure so that at the beginning of the exposures, subjects were exposed to higher concentrations than they were at the end of the exposures. During this 2-hour period, the subjects alternately exercised for 15 minutes, at a level calibrated to double minute ventilation, and rested for 15 minutes. Statistical analysis of the group average data suggested that the mixture may have been slightly more irritating to the subjects than O_3 alone. A large percentage (13 of 19) of the subjects exhibited small decrements in pulmonary function. The group average $FEV_{1.0}$ on the exposure day was statistically significantly depressed ($p < 0.001$) by 3.7 percent of the control value. One might expect O_3 alone to depress $FEV_{1.0}$ by approximately 2.8 percent under similar exposure conditions.

Kerr et al. (1981) investigated the respiratory effects associated with exposure to low levels of sulfuric acid (H_2SO_4) aerosol. Twenty-eight normal subjects were exposed (1 or 2 subjects/chamber) for 4 hours to 100 $\mu g/m^3$ H_2SO_4 aerosol of particle size 0.1 to 0.3 μm (MMD = 0.14 μm; σg = 2.9) in an environmentallly controlled exposure chamber. Over this four-hour exposure period the acidity of the aerosol in the chamber was not monitored as a function of time; however, based on the results of Kleinman et al. (1981), it appears likely that partial neutralization of the chamber atmosphere did occur. The degree of this neutralization is dependent, however, on the number of subjects in the chamber during the exposure and chamber flow rates. At one and three hours into the study on each day, bicycle ergometer exercise was performed at a workload at 100 watts at 60 RPM for 15 minutes. Of the 28 subjects, 14 were nonsmokers and 14 were cigarette smokers. None of the subjects complained of symptoms attributable to the exposure. Measurements of pulmonary function were obtained 2 hours into the exposure, immediately following exposure and 2 and 24 hours postexposure. These measurements were compared with control values obtained at comparable hours on the previous day when the subject breathed only filtered, clean air in the chamber. No statistically significant differences in pulmonary function were observed either during the exposure, immediately after exposure or 2 and 24 hours post-exposure. Similar results, which are described below, were also observed for asthmatic subjects.

Kleinman and Hackney (1978) and Avol et al. (1979) also studied the responses of six asthmatics to sulfuric acid utilizing the same procedure as described on page 13-27. These subjects had pulmonary function test results that ranged widely from normal to abnormal. The 2-hr exposures were in environments containing 75 $\mu g/m^3$ H_2SO_4. The lung functions of the asthmatics showed no statistically significant changes. Two asthmatics, the extent of their disease state not given, exhibited increases in respiratory resistance on both exposure days. Nonetheless, it was concluded that there were no convincing adverse short-term health effects of sulfuric acid. However, the authors also noted the small size of their subject pool and recommended additional studies.

Utell et al. (1981) also evaluated the responses of seventeen non-smoking asthmatics following the protocol described on page 13-28. The exposures were to various sulfate aerosols. The data presented in the manuscript are incomplete, but those available suggest that specific airway conductance (SG_{aw}) induced by carbacol was significantly potentiated ($p < 0.01$) in asthmatics breathing H_2SO_4 and NH_4HSO_4 (each 1 mg/m^3). Low sulfate exposure (0.1 mg/m^3) produced no changes in SG_{aw}; however, the two asthmatics most responsive to the high H_2SO_4 dose via inhalation exhibited a potentiation effect of sulfuric acid (at the lower concentration) on bronchoconstriction induced by carbacol. A more extensive presentation of the data obtained by these investigators will be required before a clearer evaluation of the effects of sulfates on asthmatics can be determined.

Kleinman and Hackney (1978) and Avol et al. (1979) also included six asthmatics in their studies concerned with exposures to sulfates. The asthmatics were not studied under high humidity conditions but were exposed to high concentrations (up to 372 $\mu g/m^3$) of $(NH_4)_2SO_4$. Pulmonary functions were not modified by these exposures. However, an interesting side observation was made on the asthmatics. On their first day of exposure to NH_4HSO_4 aerosol, they exhibited worse lung functions in the preexposure measurements than they had on a control day; and their lung functions improved following the aerosol exposure. Subsequent analysis of local ambient conditions showed that these subjects arrived for the aerosol testing after a 3-day period of increased SO_2 and ozone levels during a "mild air pollution episode."

Most recently Horvath et al. (1981), using eleven male subjects, investigated the effects on pulmonary function of breathing sulfuric acid aerosol (223, 418 and 939 $\mu g/m^3$; 0.9 μm) for 120 minutes in a chamber under exercising conditions (Ve ~ 30 l/minutes). The only measurement to show an interaction across time (from preexposure to postexposure) and H_2SO_4 concentration was $FEV_{1.0}$ which was significantly decreased with exposure to 939 $\mu g/m^3$ H_2SO_4 but not at the other concentrations. Symptoms commonly reported at this latter concentration include throat irritation and dryness and cough. Horvath et al. (1981) conclude that their results support previous studies (Avol et al., 1979; Kerr et al., 1981; Sackner et al., 1978; Lippmann et al., 1981; Leikauf et al., 1981), finding relatively minor or no pulmonary function changes in humans exposed to sulfuric acid aerosols in the 0.3 to 0.5 μm size range at similar concentrations (100-1000 $\mu g/m^3$). However, they do not confirm pulmonary function and other effects reported by Amdur et al. (1952) and Sim and Pattle (1957), whose subjects were likely exposed to larger size H_2SO_4 aerosols.

13.3.2 Insoluble and Other Non-Sulfur Aerosols

It is well-known that the ambient air contains many other particulate matter species besides sulfates. The non-sulfate species include, for example, condensed organic vapors (POM), lead, arsenic, selenium, hydrogen ions, ammonium salts, and carbon as soot. Health effects associated with animal and/or human occupational studies have been addressed for POM, lead, and arsenic in separate health assessment documents. Because of ethical considerations, controlled human exposure studies on these inherently toxic and/or potentially carcinogenic compounds do not exist. However, in addition to sulfate aerosols which have been extensively studied using humans, a few investigators have conducted controlled human exposure studies using carbon or other inert dust. These studies are described below and are summarized in Table 13-4.

In a study designed to determine the efficiency of nasal dust filtration, Andersen et al. (1979) exposed 16 resting subjects (12 men, 4 women) for 5 hours to three levels of plastic dust impregnated with carbon black (2, 10, and 25 mg/m^3). The number of these particles, expressed as a percent of the total number of particles was 36, 41, 14, 17, and 2, respectively, for the aerodynamic size ranges $\leq$ 1.8, 1.9 to 5.3, 5.4 to 8.9, 9.0 to 12.4, and $\geq$ 12.5 μm. Nasal mucus flows were measured at 5 different positions in the nose and varied markedly from subject to subject and in relation to the three dust concentrations. The only statistically significant effect observed was a small but statistically significant decrease in $FEV_{1.0}$. The information reported was inadequate to support this conclusion and the approximate data presented in their Figure 3 fails to confirm their statements. These investigators also noted only slight discomfort regardless of the level of dust in the environment. Apparently some further discomfort (primarily dryness in the nose and pharynx) was reported after the exposures were terminated.

Other investigators, in addition to Andersen et al. (1979), have evaluated pulmonary effects associated with breathing particles under controlled exposure conditions. Widdecombe et al. (1962) had 9 subjects inhale 20 breaths of granulated charcoal (7 to 14 mesh). Airway conductances were measured before and 2 minutes after oral inhalation of the dust. Airway conductance and thoracic gas volume ratio decreased in every subject (increased resistance) by a mean of 41 percent ($p < 0.001$). Coughing occurred in many of the subjects, and a few had mild discomfort in the upper airways. Subcutaneous injection of atropine blocked this effect, suggesting that it is produced via a vagal reflex. The response to dust inhalation was rapid and reversible, but the effective concentration levels could not be determined from the report.

Acute physiological effects of inhaling fine particulate matter were evaluated by DuBois and Dautrebande (1958). Five normal subjects were given a large number of pulmonary function tests before and after inhalation (for 1 to 3 minutes) of small (5 to 10 mg) quantities of fine chemically inert dust particles consisting of $CaCO_3$ (~ 250 mg/m^3),* coal dust (~ 500 mg/m^3),* activated charcoal powder, aluminum powder and aerosolized India ink. The size of the particles (< 0.5 μm; mean 0.04 μm) and the amount inhaled varied depending upon the substances tested. The subjects usually showed an immediate marked increase in airway resistance and pulmonary resistance. Other pulmonary functions were inconsistently altered. The effects observed were transitory, usually disappearing within 30 minutes. Constantine et al. (1959) followed up on this study, utilizing six normal subjects and seven patients with either emphysema or asthma without acute symptoms. They used dust particles (< 0.5 μm) from a colloidal iron-water suspension (~ 16.5 mg/m^3)* and from McIntyre aluminum powder (~ 100 mg/m^3).* These inhalations resulted in an increased airway resistance, confirming the previous observations of DuBois and Dautrebande. However, in patients with chronic respiratory disease, a significant reduction in vital capacity was also found. The constricting effects of these dusts persisted for a slightly longer time than previously observed (more than 25 to 45 minutes). These results are of interest in that patients with chronic respiratory disease, asthma and emphysema exhibit reactions similar to those of normal subjects. These patients, already characterized by an increase in airway resistance, exhibit reactions to constricting aerosols more severe than normal subjects; yet the percentage of increase of the airway resistance in the seven patients studied after constricting aerosols is not greater than that of the six normal control subjects.

Similar controlled human exposure studies were carried out by Toyama (1964). Ten healthy males (20 to 35 years) inhaled 20 deep breaths of concentrated (10 mg/m^3) dust which was

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*Approximate values calculated from references.

13-4. PULMONARY EFFECTS OF AEROSOLS

Concentration[a]	Duration of exposure (mins)	Number of subjects	Oral or nasal exposure	Rest or exercise	Effects	Reference
SO_2 (1.6 - 5 ppm) NaCl(CMD = 0.22 µm)	5	13	Mask	R	Synergistic increases in airway resistance with aerosol	Toyama, 1962
SO_2 (9-60 ppm) NaCl (CMD = 0.95 µm)	5	10	Mask	R	Airway resistance greater after exposure to aerosol than to exposure to SO_2 alone	Nakamura, 1964
SO_2 (0.5, 1.0 and 5.0 ppm) NaCl (CMD = 6-8 µm)	15	9	Oral*	R	Significantly decrease in $MEF_{50\%}$ VC only at 5 ppm; however, magnitude not different from SO_2 alone	Snell and Luchsinger, 1969
SO_2 (1.1 - 3.6 ppm) NaCl 2.0 - 2.7 mg/m³ MMD = 0.25 µm	30	10	Oral*	R	No effect on pulmonary functions	Burton et al., 1969
SO_2 (1, 5, 15 ppm) NaCl 10-30 mg/m³ MMD 0.15 µm, σg = 2.3	30	12	Oral*	R	Changes in pulmonary function similar to changes due to SO_2 alone not influenced by aerosol	Frank et al., 1964
SO_2 (1 ppm) NaCl 1 mg/m³ MMD 0.9 µm σg = 2.0 µm	60	9 (asthmatics)	Oral	R	Significant decreases in $\dot{V}_{max\ 50\%}$ and $\dot{V}_{max\ 75\%}$	Koenig et al., 1980
SO_2 (1 ppm) NaCl 1 mg/m³ MMD = 0.9 µm, σg - 2.0 µm	30	8 (asthmatics)	Oral*	R & E	$\dot{V}_{max\ 50\%}$, $\dot{V}_{max\ 75\%}$, $FEV_{1.0}$ and R_t decreased significantly after aerosol	Koenig et al., 1981
Mixture of : SO_2 (0.37 ppm); O_3 (0.37 ppm) and H_2SO_4 (100 µg/m³) MMD 0.5 µm, σg = 3.0	120	19	Chamber	E	Small but statistically significant decrements in in $FEV_{1.0}$ and slight increases in the incidence of clinical symptoms	Kleinman et al., 1981
$(NH_4)_2SO_4$; 85 µg/m³ $(NH_4)HSO_4$; 100 µg/m³ (MMAD 0.4 µm; σg = 2.5-3.0 for both salts)	120	5 normals 5 ozone sensitive 6 asthmatics	Chamber	E	No changes in pulmonary function	Bell and Hackney, 1977a; Kleinman and Hackney, 1978 Avol et al., 1979

TABLE 13-4. (continued)

Concentration[a]	Duration of exposure (mins)	Number of subjects	Oral or nasal exposure	Rest or exercise	Effects	Reference
Inert plastic dust impregnated with carbon black (Xerox Toner 6R9000T) (2,10, 25 mg/m^3)	300	16	Chamber Nasal	R	No significant detrimental effect on airway or nasal mucociliary clearance. Small significant decrease in $FEV_{1.0}$	Andersen et al., 1979
Inert plastic dust impregnated with carbon black (Xerox Toner 6R90005) 2, 10 mg/m^3 plus SO_2 (1 and 5 ppm)	300	16	Chamber Nasal	R	Reductions in nasal mucus flowrate, forced expiratory flow ($FEF_{25\%-75\%}$) and discomfort related principally to SO_2. SO_2 and dust effects were, at most, additive.	Andersen et al., 1981
$NaNO_3$; NaCl (control) Both 7 mg/m^3 MMAD = 0.49 μm, σg = 1.7 RH ~ 25%; crystalline solids	16	11 normals (with acute resp. influenza infection)	Oral* Nasal	R	Significant decrease in SG_{aw} ($P < 0.005$) and V_{max} 40% ($P < 0.05$). Asymptomatic airway obstruction	Utell et al., 1980
High conc. aerosol (1 mg/m^3 each) Low conc. aerosol (0.1 mg/m^3 each) MMAD = 0.5-1.0 μm Aerosols included: $NaHSO_4$ NH_4HSO_4 $(NH_4)_2SO_4$ H_2SO_4	16	16 normals 17 asthmatics	Oral*	R	SG_{aw} induced by carbachol sig. potentiated in asthmatics at 1 mg/m^3 H_2SO_4 and NH_4HSO_4 each. $FEV_{1.0}$ sig. decreased after H_2SO_4 and NH_4HSO_4. No changes in SG_{aw} with all sulfates; but two most responsive asthmatics to high H_2SO_4 dose exhibited potentiation effect on carbachol-induced bronchoconstriction at lower H_2SO_4 level	Utell et al., 1981

[a] 0.1 ppm $SO_2 \cong 262$ μg/m^3 1.0 ppm $\cong$ 2620 μg/m^3 10 ppm $\cong$ 26,200 μg/m^3

0.5 ppm $SO_2 \cong 1310$ μg/m^3 5.0 ppm $\cong$ 13,100 μg/m^3 50 ppm $\cong$ 131,000 μg/m^3

*Mouthpiece

similar in composition to fly ash and coarse-mode fugitive dust (64 percent crustal material, 10 percent sulfate, 19 percent volatile). The experimental dust (0.5 to 10 μm, 2.0 MMD) was generated, using essentially the techniques of DuBois and Dantrebande, from dust particles collected by deposition in Kawasaki City, an industrial area of Japan. Dust-induced changes in airway resistance showed a wide range of response from a postexposure decrease of a minus 18 percent to a considerable increase of up to 73 percent. The majority of subjects (8 out of 10), however, showed increses between 11 to 73 percent. The authors believe that the response appeared to be initiated by mechanical irritation rather than chemical action of the particle, since isoproterenol (a bronchodilator) abolished changes in airway resistance. Additional experiments reported as being done on 8 subjects exposed sequentially to SO_2 and dust particles (and showing potentiating effects of the two pollutants) are difficult to interpret because the experiments were not designed to account for potential carryover effects from one exposure to another.

McKerrow (1964) discussed in more detail earlier findings reported by McDermott (1962). Six subjects were exposed to four concentrations of coal dust (9, 19, 33, and 50 mg/m^3) for four hours at a time on two occasions. Responses, as determined by changes in airway resistance (R_{aw}), occurred early in the exposure to all but 9 mg/m^3 concentration, and it was concluded that "significant increases in airway resistance occurred and the response was correlated to the weight of coarse particles between 3.6 and 7 μm."

Norris and Bishop (1966) exposed normals, asthmatics and bronchitics to calcium carbonate dust. A "dust cloud highly aggregated and containing about 120,000 particles/ml (~ 21 to 170 mg/m^3; 0.5 to 5.0 μm)* and about 40,000 particles/ml (~ 0.45 to 7 mg/m^3; 0.2 to 0.5 μm)"* was breathed through a mouthpiece. The volume or mass of dust inhaled was not precisely determined. Arterial blood samples were obtained so that $A\text{-}aD_{O2}$ (alveolar-arterial difference in O_2 partial pressure) could be measured. They used dust particles (< 0.5 μm) from a colloidal iron-water suspension (~ 16.5 mg/m^3)* and from McIntyre aluminum powder (~ 100 mg/m^3).* These inhalations resulted in increased airway resistance, confirming the previous observations of DuBois and Dautrebande. However, in patients with chronic respiratory disease, a significant reduction in vital capacity was also found. The constricting effects of these dusts persisted for a slightly longer time than previously observed (more than 25 to 45 minutes). These results are of interest in that patients with chronic respiratory disease, asthma and emphysema exhibit reactions similar to those of normal subjects. These patients, already characterized by an increase in airway resistance, exhibit reactions to constricting aerosols more severe than normal subjects; yet the percentage of increase of the airway resistance in the seven patients studied after constricting aerosols is not greater than that of the six normal control subjects.

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*Approximate values calculated from references.

Utell et al. (1980) studied the effects of exposure to nitrate on the airways of 11 subjects following a respiratory infection [influenza A (H_1N_1) infection]. Each subject, under a double blind randomization protocol, orally (mouthpiece) breathed either sodium chloride (NaCl) or sodium nitrate ($NaNO_3$) for an initial period of 16 minutes and then the other aerosol for 16 minutes three hours later. The mass median aerodynamic diameter (MMAD) of the $NaNO_3$ aerosol was 0.49 µm (concentration 7,000 µg/m^3). Exposure to the nitrate aerosol (compared to NaCl aerosol) resulted in significant decreases in specific airway conductance ($p < 0.005$) and partial expiratory flows at 40 percent of TLC ($p < 0.05$) at the first two test times (48 hours and one week postinfection). They concluded that the constriction observed was a specific effect of the sodium nitrate rather than a nonspecific response to the particles.

Some experimenters have attempted to determine physical or chemical characteristics of atmospheric particles that may affect their patterns of respiratory tract deposition, interactions with other pollutants such as SO_2, and ultimately, their relative potential for contributing to toxic effects under various conditions. For example, in a physical system, Corn and Cheng (1972) found that insoluble particulate samples of $CaCO_3$, V_2O_5, and fly ash from a coalburning power plant were essentially inert to SO_2; but Fe_2O_3, MnO_2, activated carbon and suspended particulate matter from urban Pittsburgh air sorbed SO_2. They suggested that pulmonary flow resistance (measured in their earlier guinea pig studies) consequent to SO_2-aerosol exposure is sensitive to the chemical reaction product of SO_2 with specific particulate compounds.

Stahlhofen et al. (1980) determined the deposition of particles in subjects breathing at the slow rate of 7.5 breaths per minute. When the total volume was 1 liter, pulmonary deposition of 3.5 µm particles was as high as 70 percent. The deposition of various other particles (such as Fe_2O_3, di-2-ethylhexyl sebacate) have also been studied, but these are not commonly found in the environment. Additional basic information on respiratory system deposition of respirable dust particles is presented in Chapter 11.

13.4 PARTICULATE MATTER AND SULFUR DIOXIDE

As suggested by Amdur (1969) and the above Corn and Cheng (1972) study findings, one of the most significant factors influencing physiological responses to SO_2 may be the presence of certain specific kinds of particulate matter in the atmosphere. Controlled human exposure studies addressing this issue are also summarized in Table 13-4. Particulate matter appears to interact with SO_2 in at least two distinct ways: as a carrier of SO_2 and as a factor in chemical reactions resulting in the conversion of SO_2 to other forms. In their carrier role, particles may adsorb SO_2 and, depending on their size, solubility, and other characteristics, transport it deep into the respiratory system (see Chapter 11, Section 11.2 for more detailed discussion of deposition).

Respiratory function effects are illustrated by the results of studies by Nakamura (1964) and Toyama (1962), who reported that sodium chloride aerosol potentiated the response of human subjects to SO_2. In Nakamura's (1964) study, 10 subjects were first exposed to NaCl aerosol (CMD = 0.95 µm; Horvath's estimate MMAD = 5.6 µm) alone for 5 minutes, allowed to recover for

10 to 15 minutes, exposed to SO_2 alone at 9 to 60 ppm (23.6 to 157 mg/m^3) for 5 minutes, allowed 20 to 30 minutes to recover, and then exposed to SO_2 and the NaCl aerosol together for 5 minutes. Airway resistance was greater after the combination exposure than after exposure to SO_2 alone (Nakamura, 1964). As noted, the combination condition always followed exposure to SO_2 alone, thus raising the possibility that the effects of the latter exposure were confounded. However, on average, the subjects' airway resistance measures returned to only 4 percent above their pre-exposure control levels, thus making it more likely the reported effects were independent of preceding conditions. Toyama (1962) also reported that 5 minute exposures to SO_2 in combination with submicrometer (0.22 μm MMD; Horvath's estimae MMAD = 0.36 μm) particles of NaCl aerosol produced synergistic increases in airway resistance in 13 subjects, even at levels as low as 1.6 ppm (4.2 mg/m^3) SO_2. There was also a linear relationship between SO_2 concentration and percentage increase in airway resistance.

On the other hand, Burton et al. (1969) were unable to demonstrate comparable effects in 10 subjects exposed to SO_2 (1.1 to 3.6 ppm [2.9 to 9.4 mg/m^3]) in combination with NaCl aerosol (2.0 to 2.7 mg/m^3; 0.25 μm MMD; Horvath's estimate MMAD 0.4 μm). There was, however, a great deal of variability within and between subjects in this study, including one or two possible "hyper-reactors" who did show effects below 3 ppm. Frank et al. (1964) studied 12 subjects who were exposed to three conditions of SO_2 and NaCl aerosols. There were six subjects in each group, but the same subjects were not evaluated under each of the three conditions. The purpose of this study was to determine whether acute changes in respiratory dynamics Rl (pulmonary flow resistance) noted to occur during SO_2 exposure were intensified by the presence of sodium chloride particles. The NaCl aerosols had a mean geometric diameter of 0.15 m (Horvath's estimate MMAD = 0.3 μm) and a concentration of 10 to 30 mg/m^3; SO_2 concentrations were 1 to 2, 4 to 6, and 14 to 17 ppm (2.6 to 5.2, 10.5 to 15.7, and 36.7 to 44.5 mg/m^3). The subjects' response to the SO_2 exposures was as previously noted, in that Rl was not affected by the lower levels of SO_2 and progressively increased at the higher levels. The only statistically significant difference ($p < 0.05$) between the effects of the gas alone and the gas-aerosol mixture was a slightly greater average increase in pulmonary flow resistance at 4 to 6 ppm (10.5 to 15.7 mg/m^3) SO_2 than under the combination condition. Addition of the NaCl aerosol resulted in changes similar to those observed for SO_2 alone.

Snell and Luchsinger (1969) also compared the effects SO_2 alone and in mixture with aerosols of either NaCl or distilled water. Nine subjects inhaled SO_2 at 0.5, 1, and 5 ppm (1.3, 2.6, and 13.1 mg/m^3) alone and in combination with aerosols for 15-minute periods separated by 15-minute control periods. For the SO_2-saline aerosol exposure, decreases in maximum expiratory flowrate ($MEF_{50\%\ VC}$) were significant ($p < 0.01$) only at 5 ppm (13.1 mg/m^3) SO_2 whereas the SO_2-distilled water aerosol exposure produced significant decreases ($p < 0.01$) at all SO_2 exposure levels, i.e., 0.5, 1, and 5 ppm (see Figures 3 and 4, Snell and Luchsinger, 1969). The authors noted that the size of the aerosol particles differed considerably, saline particles averaging around 7 μm in diameter and water aerosols averaging less than 0.3 μm in diameter (Snell and Luchsinger, 1969; Ulmer, 1974).

Koenig et al. (1980) exposed nine adolescent resting subjects (extrinsic asthmatics) for 60 minutes to either filtered air, 1 ppm (2.6 mg/m^3) SO_2 and 1 mg/m^3 of sodium chloride droplet aerosol or 1 mg/m^3 of NaCl droplet aerosol (MMD 0.9 μm; $\sigma_g = 2.0$) alone. Exposure to SO_2 alone was not performed. Total respiratory resistance (R_T), maximal flow at 50 and 75 percent of expired vital capacity (partial flow volume), $FEV_{1.0}$, and functional residual capacity were measured before, during (30 minutes), and after exposures via forced oral breathing. No statistically significant changes were found during exposures to filtered air or NaCl aerosol alone. Significant decreases ($p < 0.025$) were observed in $\dot{V}_{max\ 50\%}$ and $\dot{V}_{max\ 75\%}$ after combined SO_2-NaCl exposure, suggesting that bronchoconstriction occurred in the peripheral airways. It should be noted that $\dot{V}_{max\ 50\%}$ was significantly depressed (8 percent) only at the midpoint of exposure. However, since possible chemical reactions can occur between dissolved SO_2 and the NaCl droplets (producing sulfite, bisulfite, and hydrogen ions), the pulmonary effects observed cannot be directly attributed to either gaseous SO_2 or to the chemical substances produced.

Koenig et al. (1981) exposed 8 adolescent extrinsic asthmatics to the same conditions as in the previous study but had them also undergo a 10-minute period of moderate exercise during the exposures. Maximum flow at vital capacity 50% and 25% above residual volume ($\dot{V}_{max\ 50\%}$ VC and $\dot{V}_{max\ 75\%}$ VC) decreased 44 and 50 percent respectively from the baseline mean after the exercise. Statistically significant changes in forced expiratory volume in one sec. ($FEV_{1.0}$) and R_T (airway resistance total) were also observed, suggesting that exercise and SO_2-NaCl exposure resulted in effects on both large as well as small airways. The functional changes seen after exercise with exposure to filtered air or NaCl droplet aerosol alone were not statistically significant. Although $\dot{V}_{max\ 50\%}$ was depressed in resting subjects (extrinsic asthmatics) 8 percent ($t = 2.83$ $p < 0.025$) and 6 percent ($t = 0.38$, p = N.S), respectively, in the 1980 and 1981 studies by Koenig et al., it should be mentioned that the latter change was not significant after 30 minutes of exposure. In the 1980 study, all subjects (N = 9) decreased in $\dot{V}_{max\ 50\%}$; however, in the 1981 study some of the eight subjects increased and some decreased.

Recently, Andersen et al. (1981) exposed 16 subjects (male and female) for 5 hours in an environmental chamber to several levels of SO_2 and dust. Subjects were exposed to clean air, to combinations of SO_2 (2.6 or 13 mg/m^3) and inert plastic dust (2 or 10 mg/m^3) or to SO_2 (13 mg/m^3) and dust (10 mg/m^3) coated with vanadium. Nasal mucus flowrate, nasal airflow resistance, forced vital capacity, and subjective discomfort were measured. Reductions in nasal mucus flowrate, forced expiratory flow ($FEF_{25\ to\ 75\%}$), and discomfort were related principally to SO_2. Sulfur dioxide and dust effects were, at the most, additive without evidence of potentiation effects. Measurements were made initially in clean air and after 2 to 3 and 4 to 5 hours of exposure to the various combinations employed. Exposures were made on five consecutive days. There was no randomization of the exposures. This may have been responsible for some of the variability in their accumulated data. They relate the data from the present study to an earlier report (Andersen et al., 1974), where SO_2 exposures induced a decrease ($p < 0.01$) in nasal mucus flowrate, more pronounced at the higher SO_2 concentration. This depression of nasal mucus flowrate caused by SO_2 exposure is apparently further increased by the plastic dust in the SO_2 environment. No potentiation effects were observed. Some caution should be expressed as to these conclusions, since different subjects were utilized and the two studies were reported 7 years apart. $FEV_{1.0}$ was reduced during SO_2 exposures ($p < 0.05$), and no change occurred during dust-alone exposures. They also report that combined exposure to dust and SO_2 produced an additive effect with respect to experienced discomfort.

13.5 SULFUR DIOXIDE, OZONE, AND NITROGEN DIOXIDE

Another important issue is whether or not SO_2 interacts synergistically with other gaseous air pollutants to produce greater-than-additive effects beyond those due to exposures to SO_2 or the other pollutants alone. Several studies have addressed this issue in relation to possible combined effects of SO_2 and other common gaseous air pollutants, as discussed below and summarized in Table 13-5.

Bates and Hazucha (1973) and Hazucha and Bates (1975) exposed eight volunteer male subjects to a mixture of 0.37 ppm O_3 and 0.37 ppm (0.99 mg/m^3) SO_2 for 2 hours. Temperature, humidity, concentrations and particle sizes of ambient aerosols (if any) were not measured. Sulfur dioxide alone had no detectable effect on lung function, while exposure to ozone alone resulted in decrements in pulmonary function. The combination of gases resulted in more severe respiratory symptoms and pulmonary function changes (10 to 20% decrements) than did ozone alone. Using the maximal expiratory flow rate at 50 percent vital capacity as the most sensitive indicator, it was evident that, after 2 hours exposure to 0.37 ppm (0.99 mg/m^3) SO_2 alone, no change occurred. However, during exposure to 0.37 ppm O_3, a 13-percent reduction was observed, while exposure to the mixture of 0.37 ppm O_3 and 0.37 ppm (0.99 mg/m^3) SO_2 resulted in a reduction of 37 percent in this measure of pulmonary function. The effects resulting from O_3 and SO_2 in combination were apparent in 30 minutes, in contrast to a 2-hour time lag for exposure to O_3 alone.

Bell et al. (1977b) attempted to replicate these studies with four normal and four ozone-sensitive subjects. They showed that the $O_3 + SO_2$ mixture had a greater detrimental effect on all pulmonary function measures than did O_3 alone. However, only some of the lung function parameters measured showed statistically significant decrements when compared to O_3. Four of the Hazucha and Bates (1975) study subjects were aso studied by Bell et al. (1977b). Two of these subjects had unusually large decrements in FVC (40 percent) and FEV_1 (44 percent) in the first study (Bates and Hazucha, 1973), while the other two had small but statistically significant decrements. None of the subjects responded in a similar manner in the Bell study. Retrospective sampling of the ambient air conditions utilizing particle samplers and chemical analysis in the chamber showed that acid sulfate particles could have been 10- to 100-fold higher in the Hazucha and Bates chamber and thus might have been responsible for the synergistic effects observed; however, this hypothesis was not confirmed in recent studies conducted

13-5. PULMONARY EFFECTS OF COMBINED EXPOSURES TO SO_2 AND OTHER GASEOUS AIR POLLUTANTS

Concentration[a]	Duration of exposure (mins)	Number of subjects	Source	Effects	Reference
SO_2 (0.15 ppm) and O_3 (0.15 ppm)	120	6	Chamber (exercise)	Decrease in SG_{aw} after exposure to O_3. Synergistic potentiation reported for SO_2 + O_3, but statistics basis unclear	Kagawa and Tsuru, 1979
SO_2 (0.37 ppm) and O_3 (0.37 ppm)	120	8	Chamber (exercise)	Decreased pulmonary functions (synergistic effect of SO_2/O_3 greater than O_3 alone on FRC, $FEV_{1.0}$, MMFR, $MEFR_{50\%}$)	Hazucha and Bates, 197 Bates and Hazucha, 1973
SO_2 (0.37 ppm) and O_3 (0.37 ppm)	120	4 (normal) 4 (ozone sensitive) 4 (from Bates)	Chamber (exercise)	Unable to confirm SO_2/O_3 synergistic effects beyond pulmonary decrement due to O_3 alone	Bell et al., 1977b
SO_2 (0.40 ppm) and O_3 (0.40 ppm	120	9	Chamber (exercise)	Unable to confirm SO_2/O_3 synergistic effects beyond changes due to O_3 alone	Horvath and Folinsbee 1977; Bedi et al., 1979

TABLE 13-5. (continued)

Concentration[a]	Duration of exposure (mins)	Number of subjects	Source	Effects	Reference
SO_2 (0.3 ppm) + NO_2 (0.5 ppm)	120	19 asthmatics	Chamber (exercise)	No pulmonary alterations	Linn et al., 1980
SO_2 (0.5 ppm) + NO_2 (0.5 ppm)	120	24 normals	Chamber (exercise)	No pulmonary alterations	Linn et al., 1980
SO_2 (5 ppm) and NO_2 (5 ppm)	120	11	Chamber (exercise)	No changes in PaO_2, $PaCO_2$, pHa or TGV. Airway resistance (Raw) increased significantly.	von Nieding et al., 1979
SO_2 (5 ppm) NO_2 (5 ppm) and O_3 (0.1 ppm)	120	11	Chamber (exercise)	No changes in PaO_2, $PaCO_2$, pHa or TGV. Airway resistance (Raw) increased significantly.	von Nieding et al, 1979
SO_2 (0.12 ppm) NO_2 (0.06 ppm) and O_3 (0.025 ppm)	120	11	Chamber (exercise)	No changes in pulmonary functions	von Nieding et al., 1979

TABLE 13-5. (continued)

Concentration[a]	Duration of exposure (mins)	Number of subjects	Source	Effects	Reference
Mixture of: SO_2 (5 ppm) NO_2 (5 ppm) O_3 (0.1 ppm)	120	24	Chamber (exercise)	Data not adequately analyzed and could not be from the data presented.	Islam and Ulmer, 1979a
Mixture of: SO_2 (0.33 ppm) NO_2 (0.16 ppm) O_3 (0.075 ppm)	8 hr/d for 4 successive days	15	Chamber (rest)	Statistical analysis of the data not adequate	Islam and Ulmer, 1979b

[a] 0.1 ppm SO_2 ≡ 262 μg/m³   1.0 ppm ≡ 2620 μg/m³   10 ppm ≡ 26,200 μg/m³

0.5 ppm SO_2 ≡ 1310 μg/m³   5.0 ppm ≡ 13,100 μg/m³   50 ppm ≡ 131,000 μg/m³ by Kleinman et al. (1981) (see Section 13.3.1.2) involving identical concentrations of ozone and SO_2 and 100 μg/m^3 of sulfuric acid. In the Montreal chamber studies (Bell et al., 1977b), concentrated streams of SO_2 and O_3 exited from tubes separated by 8 inches (20 cm) under a fan which forced 167 ft^3/minute (4.7 m^3/minute) of air-conditioned laboratory air with SO_2 and O_3 through the chamber and out an exhaust line on the opposite wall. The concentrated streams of SO_2 and O_3 could have reacted rapidly with each other and with ambient air impurities like olefins, to form a large number of H_2SO_4 nuclei which grew by homogenous condensation, coagulation, and absorption of NH_3 during their 2-minute average residence time in the chamber.

Horvath's group (Horvath and Folinsbee, 1977; Bedi et al., 1979) exposed nine young men (18 to 27 years old) to 0.4 ppm O_3 and 0.4 ppm (1.0 mg/m^3) SO_2 singly and in combination for 2 hours in an inhalation chamber at 25°C and 45 percent relative humidity. The subjects exercised intermittently for one-half of the exposure period. A large number of pulmonary function tests were conducted before, during, and after the exposure. Subjects exposed to filtered air or to 0.4 ppm SO_2 showed no significant changes in pulmonary function. When exposed to either O_3 or O_3 plus SO_2, the subjects showed statistically significant decreases in maximum expiratory flow, forced vital capacity, and inspiratory capacity. There were no significant differences between the effects of O_3 alone and the combination of O_3 + SO_2; thus, no synergistic effects were discernible in their subjects. Although particulate matter was not present in the inlet air, it is not known whether particles developed in the chamber at a later point.

Chamber studies were also conducted by Kagawa and Tsuru (1979) who exposed six subjects for 2 hours with intermittent exercise (50 watts; i.e. ventilation of 25 l/min) for periods of 15 minutes exercise separated by periods of 15 minutes rest. The exposures were performed weekly in the following sequence: filtered air, 0.15 ppm O_3; filtered air, 0.15 ppm (390 μg/m^3) SO_2; and finally filtered air, 0.15 ppm O_3 - 0.15 ppm SO_2. Pulmonary function measurements were obtained prior to exposure after 1 hour in the chamber and after leaving the chamber. Although a number of pulmonary function tests were performed, change in specific airway conductance (SG_{aw}) was utilized as the most sensitive test of change in function. They reported a significant decrease in five of the six subjects exposed to O_3 alone. In three of the six young male subjects, they reported a significantly enhanced decrease in SG_{aw} after exposure to the combination of pollutants compared to the decrease in SG_{aw} in these subjects with O_3 exposure alone. Two other subjects had similar decreases in either O_3 or O_3 + SO_2 exposure. Kagawa and Tsuru (1979) suggested that the combined effect of the two gases on SG_{aw} is more than simply additive. Questions concerning the statistical approach employed by them, however, argue for caution in accepting their conclusions regarding greater-than-additive effects. Also subjective symptoms of cough and bronchial irritation were reported to occur in subjects exposed to O_3 or the O_3 + SO_2 combination. The question of potential synergistic interaction between SO_2 and O_3, therefore, remains unresolved by this study.

Recent studies by Bedi et al. (1981), under conditions of high temperature and humidity, attempted to explain the conflict of opinion as to the cause of synergistic effects reported by Hazucha and Bates (1975) and Kagawa and Tsuru (1979) for humans exposed to the combined gases of O_3 and SO_2. While exercising (Ve ~ 30 l/minutes), eight young nonsmoking adult males were randomly exposed for 2 hours to filtered air, 0.4 ppm (1.1 mg/m^3) SO_2, 0.4 ppm O_3 and 0.4 ppm SO_2 plus 0.4 ppm O_3 at 35°C and 85% RH. No functional changes in $FEV_{1.0}$ occurred as a result of exposure to filtered air or 0.4 ppm (1.1 mg/m^3) SO_2, but significant decreases occurred following exposure to either 0.4 ppm O_3 (6.9%) or the combination of 0.4 ppm O_3 plus 0.4 ppm (1.1 mg/m^3) SO_2 (7.4%). However, no significant differences were found between the ozone exposure and the ozone plus sulfur dioxide exposure. Observed alterations in pulmonary functions in the SO_2 + O_3 exposure reflects the changes occurring during the O_3 exposure, clearly indicating that no synergistic effects related to the additional presence of SO_2 were evident in this study.

In another recent study, Linn et al. (1980) exposed 24 normal subjects to clean air and to a combination of 0.5 ppm (0.9 mg/m^3) NO_2 and 0.5 ppm (1.3 mg/m^3) SO_2 for two hours. The exposure was conducted at an ambient temperature of 31°C dry bulb and 40% relative humidity. Fifteen-minute exercise periods requiring a ventilatory exchange of approximately 30 liters/minute were alternated with 15-minute rest periods. Group mean lung function was essentially similar in both exposure conditions with no changes occurring consequent to the exposures. There was a small, statistically significant increase in symptoms during and after the exposure. (Follow-up postexposure symptomatology reports were obtained by telephone.) Linn et al. (1980) also exposed 19 asthmatic subjects (of widely different clinical status) to 0.5 ppm (0.9 mg/m^3) NO_2 and 0.3 ppm (0.8 mg/m^3) SO_2 as well as to filtered air following a protocol similar to that conducted in their concurrent studies of normal subjects. Again no pulmonary alterations were observed, a finding similar to that obtained for normal subjects. Also, no subjective symptoms were observed during the exposure, but small, statistically significant increases in symptoms were reported during the telephone followup later in the day. These changes were not necessarily attributable to the earlier test exposure. Overall, then, the Linn et al. (1980) study does not provide evidence for synergistic interactions between SO_2 and NO_2 at the concentrations studied, for either normal or asthmatic subjects.

Von Nieding et al. (1979) exposed 11 subjects to O_3, NO_2 and SO_2 singly and in various combinations. The subjects were exposed for 2 hours with 1 hour devoted to exercise that doubled their ventilation. The work periods were of 15-minute duration interspersed with 15-minute periods at rest. In the actual exposure experiments, no significant alterations were observed for O_2, CO_2, and pH in arterialized pulmonary blood and thoracic gas volume (TGV). Total airway resistance (R_T) and arterial oxygen tension (PaO_2) were altered in certain studies. Arterial oxygen tension was decreased (7 to 8 torr) by exposure to 5.0 ppm (9.2 mg/m^3) NO_2 but was not further decreased following exposures to 5.0 ppm (9.2 mg/m^3) NO_2 and

5.0 ppm (13.1 mg/m^3) SO_2 or to 5.0 ppm (9.2 mg/m^3) NO_2, 5.0 ppm (13.1 mg/m^3) SO_2, and 0.1 ppm O_3 or 5.0 ppm (9.2 mg/m^3) NO_2 and 0.1 ppm O_3. Airway resistance increased significantly [0.5 to 1.5 cm H_2O/(L/s)] in the combination experiments to the same extent as in the exposures to NO_2 alone. In the 1-hour post exposure period of the NO_2, SO_2, and O_3 experiment, R_T continued to increase. Subjects were also exposed to a mixture of 0.06 ppm (110 µg/m^3) NO_2, 0.12 ppm (310 µg/m^3) SO_2, and 0.025 ppm O_3. No changes in any of the measured parameters were observed. These same subjects were challenged with a 1, 2, and 3 percent solution of acetylcholine following control (filtered air) exposure and to the 5.0 ppm (9.2 mg/m^3) NO_2, 5.0 ppm (13.1 mg/m^3) SO_2, and 0.1 ppm O_3 mixture, as well as after the 0.06 ppm (110 µg/m^3) NO_2, 0.12 ppm (310 µg/m^3) SO_2, and 0.025 ppm O_3 mixture exposures. Individual pollutant gases were not evaluated separately. The expected rise in airway resistance was observed in the control study. Specific airway resistance (R_{aw} x TGV) was significantly greater than in the control study following the combined pollutant exposures.

In another study of simultaneous exposure to SO_2, NO_2 and O_3, three groups of eight subjects, each of different ages (<30, >49 and between 30-40 years) were exposed for 2 hours in a chamber on three successive days (Islam and Ulmer, 1979a). On the first day, subjects breathed filtered air and exercised intermittently (levels not given); on the second day they were exposed at rest to 5.0 ppm (13.1 mg/m^3) SO_2, 5.0 ppm (9.2 mg/m^3) NO_2 and 0.1 ppm O_3; on the third day the environment was again 5.0 ppm (13.1 mg/m^3) SO_2, 5.0 ppm (9.2 mg/m^3) NO_2 and 0.1 ppm O_3 but the subjects exercised intermittently during the exposure. Statistical evaluation of the data for 11 lung function test parameters and two blood parameters (PaO_2 and $PaCO_2$) were not adequately performed. These measurements were made before, immediately, and 3 hours post exposure. Individual variability was quite marked. The investigators concluded that no synergistic effects occurred in their healthy subjects. However, since they did not systematically expose these subjects to the individual components of their mixed pollutant environment, the conclusion can only be justified in that they apparently saw no consistent changes. There were some apparent changes in certain individuals related to exercise (unknown level) and age, but the data were not adequately analyzed nor could they be, based on the information presented.

Islam and Ulmer (1979b) studied 15 young healthy males during chamber exposures to 0.9 mg/m^3 SO_2, 0.3 mg/m^3 NO_2 and 0.15 mg/m^3 O_3. Ten subjects were exposed to 1 day of filtered air and 4 succesive days to the above gas mixture. Another group of 5 subjects were exposed for 4 days to the pollutant mixture followed by 1 day to filtered air. Each exposure was 8 hours in duration. Following each exposure, the subjects were challenged by an acetylcholine aerosol. Eight pulmonary function tests and four blood tests (PaO_2, $PaCO_2$, Hb, and lactate dehydrogenase) were performed before and after the exposure. No impairments of lung functions, blood gases or blood chemistry were found, but the study suffers from a deficiency in statistical analysis of the data. Also, some of the subjects were said to have exhibited unusual responses.

13.6 SUMMARY AND CONCLUSIONS

Unlike community epidemiological studies that investigate health responses of large population cohorts under highly variable ambient exposure conditions, controlled human exposure (clinical) studies typically evaluate much smaller numbers of subjects but under much better defined and carefully controlled exposure conditions. In the latter type of studies, exposures to either single pollutants or combinations of pollutants are usually carried out in environmentally controlled chambers in which relative humidity, temperature, and pollutant concentrations are designed to approximate representative ambient air exposure conditions, especially those thought to be associated with the induction of acute effects.

Generally inherent in the design of controlled human exposure studies carried out in the United States are limitations on the range or types of pollutant exposures and types of subjects studied so as to assure (as approved by human rights and medical ethics committees) that the experimental exposures to the pollutants being tested per se will not lead to serious morbidity, irreversible illness, or death. Consequently, the types of pulmonary responses typically assessed in controlled exposure studies are typically "transient" and "reversible." However, depending upon the population at risk, the method of exposure, and the level of subject activity, the so-called mild and reversible health effects measured in controlled human exposure studies may be indicators of other more serious associated health effects likely to occur if more prolonged or repeated ambient exposures to the same concentrations of pollutants were encountered by study subjects; or the observed effects per se may be sufficient to interfere with normal work or social activities of certain individuals under some ambient circumstances. For example, relatively small increases in airway resistance of no particular health concern for healthy, nonsensitive adults may be of medical importance for asthmatics or other sensitive groups with already compromised pulmonary functions, especially when accompanied by symptoms associated with or indicative of the onset of more severe breathing difficulties for them under ambient conditions.

In general, the population groups at special risk to air pollution include the young, the elderly, and individuals predisposed by some particular disease, including asthma, bronchitis, cystic fibrosis, emphysema, and cardiovascular disease. In the normal population, there are also nondiseased but hypersensitive individuals. Such nondiseased "hyperreactors" have been found among at least three of the distinct population groups (normals, chronic bronchitics, and asthmatics) that have been evaluated under controlled exposure conditions in regard to their responses to SO_2 and particulate matter (Lawther, 1955; Frank et al., 1964; Nadel et al., 1965; Burton et al., 1969; Lawther et al., 1975; Jaeger et al., 1979; Sheppard et al., 1980, 1981; Stacy et al., 1981).

In evaluating responses of the above population groups, various investigators have assessed the effects of varying the activity levels of the subjects, the mode of exposure (e.g., nasal, oral, oronasal, or open chamber), and the duration of exposure. One purpose of increasing the activity level during exposure is to increase minute ventilation (Ve), so as to simulate outdoor exposures during daily activities. A large majority of normal subjects at rest breathe almost exclusively through the nose with a minute volume (Ve) of approximately 5 to 10 liters/minute. However, some healthy individuals may have abnormally obstructed nasal passages, or, for other reasons, regularly breathe oronasally even at rest, as indicated by recent studies (Niinimaa, 1980, 1981; D'Alfonso, 1980), and certain population groups at risk (such as asthmatics) include some individuals who tend to breathe orally even at rest. At some level of increased ventilation, individuals who normally breathe through the nose at rest also shift over to oronasal breathing. In regard to ventilation levels at which that shift has been observed to occur, Niinimaa et al. (1980, 1981) reported a switch from nasal to oronasal breathing at a minute volume of 35.3 ± 10.8 (mean ± S. D.) liters per minute, and after the switch to oronasal breathing by persistent nasal breathers (at rest), the nasal portion of Ve decreased to 56 percent of total Ve. In addition to the study recently published by Niinimaa et al. (1980, 1981), D'Alfonso (1980) also observed the shift to oronasal breathing in response to increasing ventilation rate and found that subjects who are nasal breathers at rest move to oronasal breathing at a mean minute volume (Ve) of 30 liters per minute. At maximum exercise levels (90 liters/minute), subjects breathe, at most, 40% of the total minute volume through the nose.

The results of such studies are extremely important in aiding our understanding of results reviewed here as being derived from controlled human exposure studies of PM and SO_2. Sulfur dioxide, for example, is very soluble in water and, when inhaled nasally, is readily (95 to 99 percent) absorbed on the moist surfaces of the nose and upper respiratory passages (Frank et al., 1973). This, in fact, may protect individuals breathing nasally at rest from even relatively high levels of SO_2 exposure. At some level of ventilation, however, breathing shifts from nasal to oronasal, thereby increasing the dose of SO_2 reaching the tracheobronchial region of the lung and probably leading to enhanced SO_2 effects at ambient exposure levels below those affecting the same individuals while breathing nasally at rest or at lower activity levels. Forced oral breathing yields less nasopharyngeal absorption than either nasal or oronasal breathing and would be expected to yield a more intense exposure-effect relationship than is observed with either nasal or oronasal breathing. As summarized below, these expected patterns can be discerned clearly when examining the results of available controlled human exposure studies, especially in regard to SO_2 effects.

13.6.1 Sulfur Dioxide Effects

Sulfur dioxide has been found to affect a variety of physiological functions. These include sensory processes, subjective perceptions of irritative or painful SO_2 effects, and more objectively measured changes in respiratory function parameters. Although the reliability of subjective reports of perceived effects of SO_2 has been questioned by some, certain statements can be made with confidence concerning SO_2 effects on sensory processes. For example, exposure to 5 ppm of SO_2 results uniformly in the detection of the odor of SO_2, while odor detection below that level varies considerably. Other changes (e.g., alterations in electroencephalogram alpha rhythms or an impact on response of the dark adapted eye to light) have been reported to occur at SO_2-exposure levels as low as 0.20 to 0.23 ppm. However, the health significance of such "sensory effects" is unclear at this time, but would appear to be of relatively little concern unless any resulting discomfort or other outcome would markedly alter normal activities of affected subjects.

Of much more concern are cardiovascular or respiratory effects found to be associated with exposure to SO_2. For healthy subjects at rest, in general, such effects have not been consistently observed except at exposure levels above 5 ppm (13.1 mg/m^3). These include, for example, observations by Frank et al. (1962) of marked pulmonary flow resistance increases (mean = 39%) at 5 ppm (13.1 mg/m^3) and consistent observations by numerous other investigators listed in Table 13-2 of increased airway resistance or other bronchoconstrictive effects with exposures of healthy adult subjects to SO_2 levels of 5 ppm (13.1 mg/m^3) or higher. Only Amdur et al. (1958) has reported observations of significant cardiorespiratory effects in healthy adults at rest following SO_2 exposures below 5 ppm (13.1 mg/m^3), including exposures as low as 1 ppm (2.6 mg/m^3). Other investigators (e.g., Lawther, 1955; Frank et al., 1962) have not observed similar results in attempting to replicate the findings of Amdur et al. (1958) at levels below 5 ppm (13.1 mg/m^3). Numerous explanations could be offered for this apparent discrepancy in reported exposure-effect relationships for bronchoconstrictive effects in healthy adults at rest, but no clear resolution of the issue is presently available. Nevertheless, available evidence points to 5.0 ppm (13.1 mg/m^3) as being the most probable lowest observed effect level for induction of bronchoconstriction effects in healthy adults exposed to SO_2 while at rest.

Probably of more crucial importance are the findings of several investigators suggesting potentiation of SO_2 airway effects in normal subjects as the result of increased oral inhalation of SO_2, due either to forced mouth breathing or increased exercise levels or both. As indicated in Table 13-2, for example, deep breathing of SO_2 at 1 ppm (2.6 mg/m^3) increased SR_{aw} significantly in comparison to breathing air alone (Lawther, 1975). Also, Melville (1970) reported greater decreases in SG_{aw} with oral breathing than nasal breathing at 2.5 ppm (6.6 mg/m^3) SO_2; and Snell and Luchsinger (1969) found significant decreases in $MEF_{50\%}$ at 1 ppm (2.6 mg/m^3) SO_2 with oral breathing at rest but not at 0.5 ppm (1.3 mg/m^3) SO_2. Similarly, Jaeger et al. (1979) observed no pulmonary effects in resting normal subjects with forced oral breathing at 0.5 ppm (1.3 mg/m^3) SO_2. These studies suggest possible bronchoconstriction effects in healthy adults with oral breathing of 1.0 to 2.5 ppm (2.6 to 6.6 mg/m^3) SO_2, raising the possibility of such effects being seen at similar concentrations in healthy adults exercising at sufficient workloads to induce a shift to oronasal breathing.

Examining the effects of exercise, Kreisman et al. (1976) found that light exercise potentiated the effect of SO_2, with $MEF_{40\%}$ being significantly decreased with exercise during oral exposure of normal subjects to 3 ppm (7.9 mg/m^3) SO_2 or above. Another study, by Bates and Hazucha (1973), reported a 20 percent (but not statistically significant) decrease in MEFR with 0.75 ppm (2.0 mg/m^3) exposure of exercising adults in an open chamber; and Stacy et al. (1981) reported slight but statistically significant SR_{aw} increases in healthy adults exposed to 0.75 ppm (2.0 mg/m^3) SO_2 while exercising in a controlled exposure chamber. These effects were the only significant ones found from among numerous pulmonary function tests even under rather extreme exercise conditions employed in the Stacy et al. (1981) study. These results (Bates and Hazucha, 1973; Stacy et al., 1981), therefore, provide only very weak evidence for effects in exercising healthy adults at SO_2 levels <1.0 ppm (2.6 mg/m^3). In other studies, no pulmonary effects were observed with chamber exposures of exercising healthy adults at SO_2 exposure levels of 0.50, 0.40, or 0.37 ppm (1.31, 1.05, or 0.97 mg/m^3) (Horvath and Folinsbee, 1977; Bedi et al., 1979; Bates and Hazucha, 1973; Hazucha and Bates, 1975; Bell et al., 1977b; Linn et al., 1980; Bedi et al., 1981). The weight of available evidence, therefore, appears to indicate that induction of pulmonary mechanical function effects may occur at $\geq$ 1 to 3 ppm (2.6 to 7.9 mg/m^3) SO_2 in exercising healthy adults but not at $\leq$ 0.50 ppm (1.31 mg/m^3) SO_2 even with exercise or forced oral breathing.

In attempting to define populations at special risk for SO_2 effects, Weir and Bromberg (1972) and Reichel (1972) exposed patients with obstructive pulmonary disease to SO_2 levels across the range of 0.3 to 4.0 ppm (0.8 to 10.5 mg/m^3) and observed no statistically significant increase in airway resistance or other pulmonary function effects. The exposures were carried out while the subjects were at rest in a controlled exposure chamber, but no assessment was conducted regarding possible enhanced effects of increased oral inhalation due to exercise or forced mouth breathing. Thus, although no evidence was obtained for increased susceptibility of these patients at rest, possibly enhanced vulnerability to SO_2 effects of such subjects at elevated activity levels cannot be ruled out based on the reported results.

A clearer picture of probable enhanced susceptability or special risk for SO_2-pulmonary function effects appears to be emerging now in regard to asthmatic subjects. For example, Jaeger et al. (1979) reported observing small, statistically significant (mean 2.7%) decreases in MMFR levels (which recovered in 30 minutes) following forced oral exposure (by use of nose clips) to 0.5 ppm (1.3 mg/m^3) SO_2 of 40 asthmatic subjects at rest in a controlled exposure chamber. Two subjects experienced delayed effects requiring medication that may have been due to the SO_2 exposures. (Other uncontrolled factors, however, cannot be ruled out as possibly having caused the delayed symptoms.) While the small pulmonary function decrements observed by Jaeger et al. (1979) may be physiologically insignificant per se, they are suggestive of possible SO_2 effects occurring in asthmatics at SO_2 levels below those affecting nonsensitive healthy adults.

Consistent with this possibility, Sheppard et al. (1980) observed statistically significant SR_{aw} increases in clinically defined mild asthmatics with oral exposures to 1, 3, or 5 ppm (2.6, 7.7 or 13.1 mg/m^3) SO_2 via mouthpieces while at rest but observed significant SR_{aw}

increases in normal and atopic subjects only at 5 ppm (13.1 mg/m^3). In further studies, Sheppard et al. (1981) observed statistically significant increases in SR_{aw} with oral exposure of asthmatics to 0.25 and 0.5 ppm (0.7 and 1.3 mg/m^3) SO_2 via forced mouth breathing while exercising at moderately elevated level (Ve ≅ 30 liters/minute). The two most responsive subjects of six tested experienced increased SR_{aw} with oral exposure to levels as low as 0.10 ppm (260 μg/m^3) SO_2. At 0.5 ppm three of the subjects experienced wheezing and shortness of breath, and at 1.0 ppm all six subjects experienced such symptoms. Sheppard et al. (1980) also employed pharmacologic tests, which indicated that the very rapid onset bronchoconstrictive effects seen in the asthmatics are under parasympathetic neural control, as was earlier demonstrated (Nadel et al., 1965) to be the case for normal subjects experiencing bronchoconstriction in response to exposure to SO_2 at a higher level (i.e., 5 ppm) while at rest.

The Sheppard et al. (1980, 1981) results appear to demonstrate that some asthmatic subjects may be approximately an order of magnitude more sensitive to SO_2 exposure than normal, nonsensitive healthy adults. That is, whereas nonsensitive healthy adults display increased bronchoconstriction at 5 to 10 ppm while at rest and at levels possibly as low as 1 ppm with oral or oronasal breathing, clinically defined mild asthmatics appear to be sensitive, as a group, down to 0.25 ppm SO_2 and the most sensitive (as individuals) possibly down to 0.1 ppm under moderate exercise (Ve ~ 30 liters/minute) conditions. Most importantly, with brief 10 minute exposures to SO_2 concentrations encountered in U.S. cities (0.1 to 0.5 ppm), Sheppard et al. (1981) demonstrated that moderate exercise increased the bronchoconstriction produced by SO_2 in subjects with mild asthma. The results were qualitatively confirmed by Linn et al. (1982) using techniques similar to those employed by Sheppard et al. (1981). In this pilot study five asthmatics were exposed, via mouthpiece, to 0.5 ppm SO_2 for a period of 10 minutes while exercising (~ 400 kg-m/min). Similar results using face mask have been recently described (see Appendix, Chapter 13, for an abstract summarizing pertinent research recently completed and described in a manuscript submitted for peer-review and publication). However, caution should be employed in regard to any attempted extrapolation of these observed quantitative exposure-effect relationships to what might be expected under ambient conditions. Additional research results from studies using open chamber oronasal breathing conditions more analogous to those encountered in daily activities have recently been described by Linn et al. (1982). In this large-scale chamber study employing 24 asthmatic subjects, no statistically significant pulmonary function decrements were found with 0.5 ppm SO_2 exposures for 1 hour under intermittent exercising conditions. These negative results are in contrast to the findings of Sheppard et al. (1981) and Linn et al. (1982) obtained with 0.5 ppm SO_2 exposure via mouthpiece while exercising. These differences may be due to the delivery of a higher proportion of inhaled SO_2 to the tracheobronchial and lung regions with mouthpiece exposure or to individual variations in bronchial reactivity to SO_2 among subjects used in the different studies.

The health significance of pulmonary function changes and associated symptomatic effects demonstrated to occur in response to SO_2 by the above human exposure studies is an important issue for present air quality criteria development purposes. In contrast to the sensory effects of SO_2 earlier described as probably being of little health significance, much more concern is generally accorded to the potential health effects of pulmonary function changes (such as increased bronchoconstriction) and associated symptomatic effects (such as coughing, wheezing and dyspnea or shortness of breath) observed with human exposures to SO_2, especially in sensitive population groups such as asthmatics. Temporary, small decrements in pulmonary airway functions observed in some of the above studies for nonsensitive healthy adults at SO_2 concentrations ≅ 1 to 5 ppm are generally of less concern in terms of their implications regarding the potential health impact of ambient air SO_2 exposures than are the pulmonary function and symptomatic effects observed in mild asthmatics at similar (1 to 5 ppm) or lower (< 1 ppm) concentrations of SO_2. Probably of most concern are marked increases (> 10 percent) in airway resistance and symptomatic effects (wheezing, dyspnea) observed by Sheppard et al. (1981) in a group of mild asthmatics with oral exposure via mouthpiece to 0.5 ppm (1.3 mg/m^3) sulfur dioxide during exercise. A recent article (Fischl et al., 1981) and accompanying editorial (Franklin, 1981) in the medical literature discuss the inclusion of indices of airway obstruction and presenting symptoms such as wheezing and dyspnea among factors to be considered in attempting to predict the need for hospitalization of asthma patients following initial emergency room treatment (e.g., bronchodilator therapy, etc.) for asthmatic attacks.

Particulate matter, especially hygroscopic salts, have been shown to be potentially important in enhancing the pulmonary function effects of SO_2 exposure. Airway resistance increased more after combined exposure to SO_2 and sodium chloride in several studies, although others have failed to demonstrate the same effect. This difference in response to the SO_2-NaCl aerosol mixtures may be due principally to the relative humidity at the time of the exposure. McJilton et al. (1976) have demonstrated in guinea pigs that changes in pulmonary mechanical function were seen only when the mixture (SO_2/Nacl) was administered at high relative humidity (r.h. > 80%). The effect is ascribed to absorption of the highly soluble SO_2 into the droplet before inhalation, whereas at a r.h. < 40% the aerosol was a crystal. Significant reduction in $MEF_{50\%}$ VC (maximal expiratory flowrate) was observed for the group-mean after oral exposure to a combination of saline aerosol and 13.3 mg/m^3 (5 ppm) SO_2; however, no effects were observed at SO_2 levels of 0.5 ppm and 1.0 ppm (1.3 mg/m^3 and 2.6 mg/m^3) (Snell and Luchsinger, 1969). The validity of this study has been questioned based on the lack of an air sham control group and also based on the methodology used to measure $MEF_{50\%}$ VC. More recently, studies have been reported showing pulmonary function changes in extrinsic asthmatics both at rest (Koenig et al., 1980) and during exercise (Koenig et al., 1981) with exposure to 2.62 mg/m^3 (1 ppm) SO_2 and 1 mg/m^3 NaCl. Statistically significant decreases in $V_{max\ 50\%}$ and $V_{max\ 75\%}$ were observed both at rest and during exercise for asthmatics but not for all normals. Although NaCl alone produced no such effects, the lack of an "SO_2 alone" group and the difference in the number of subjects used with NaCl alone and in combination with SO_2 make interpretation difficult.

In contrast to the apparent enhancement of SO_2-induced pulmonary airway effects by combined exposure with certain particulate matter aerosols, there is less evidence that supports the hypothesis that synergistic interactions between SO_2 and other gaseous pollutants, such as O_3 or NO_2, produce greater-than-additive effects of each individually on pulmonary mechanical functions.

Controlled human exposure study evidence regarding SO_2 effects on respiratory defense mechanisms, such as mucus clearance processes, is highly limited at present. For healthy adults exposed to SO_2 while at rest, nasal mucus flowrate appeared to decrease markedly (by 50 percent) at 5.0 ppm (13.1 mg/m^3) SO_2 (Andersen et al., 1977), but tracheobronchial mucociliary clearance appeared to be unaffected by exposure at the same SO_2 level while at rest (Wolff et al., 1975a). These observed differences may be due to the much greater dose of SO_2 delivered to nasal passages than to tracheobronchial regions by nasal breathing at rest. Oral exposure of healthy adults to 5.0 ppm (13.1 mg/m^3) SO_2 during exercise (which notably increases tracheobronchial deposition of SO_2), however, was observed in two studies (Wolff et al., 1975b; Newhouse et al., 1978) to increase tracheobronchial clearance rates. No studies, to date, have investigated whether or not repeated exposures to 5.0 ppm (13.1 mg/m^3) SO_2 would continue to induce increased nasal or tracheobronchial clearance or, possibly, cause eventual slowing of mucus clearance. (Note that one early study by Cralley [1942] reported decreased mucociliary activity in healthy adults exposed to high (> 15 ppm) SO_2 concentrations while at rest.) Nor have any controlled exposure studies investigated the effects of SO_2 exposure on mucus clearance activities in asthmatics or other potentially sensitive human population groups, such as individuals with chronic obstructive pulmonary diseases. Thus, while SO_2 effects on nasal and tracheobronchial mucus clearance processes cannot be said to have been demonstrated to occur in sensitive population groups at exposure levels below those affecting healthy adults, such a possibility cannot be ruled out at this time.

13.6.2 Sulfuric Acid and Sulfate Effects

In addition to SO_2 being absorbed by hygroscopic particles, whereby its effects may be potentiated, sulfur dioxide is also transformed into sulfur trioxide during transport and (in combination with moisture) sulfuric acid is formed. The latter may exist as a sulfuric acid droplet or can be converted to sulfates in the presence of ammonia, which is found in the ambient air and in expired human breath.

Sulfuric acid and other sulfates have been found to affect both sensory and respiratory function in study subjects. The odor threshold for sulfuric acid has been estimated to be at 0.75 mg/m^3 based on one study and at 3.0 mg/m^3 based on another.

Respiratory effects from exposure to sulfuric acid mist (0.35 to 0.5 mg/m^3) have been reported to include increased respiratory rate and decreased maximal inspiratory and expiratory flowrates and tidal volume (Amdur et al., 1952). However, several other studies of pulmonary function in nonsensitive healthy, adult subjects (Newhouse et al., 1978; Sackner et al., 1978; Kleinman et al., 1978; Avol et al., 1979; Leikauf et al., 1981; Kerr et al., 1981; Horvath et al., 1981) indicated that pulmonary mechanical function was little affected when subjects were exposed at 0.1 to 1.0 mg/m^3 sulfuric acid for 10 to 120 minutes, although in one study (Utell et al., 1981) the bronchoconstrictor action of carbachol was potentiated by the sulfuric acid and sulfate aerosol, more or less in relation to their acidity.

In regard to mucoci[illegible]learance effects, tracheobronchial clearance was significantly increased at 100 μg/m^3 H_2S[illegible]$_4$, was not significantly altered at 300 μg/m^3, but significantly decreased at 1000 μg/m^3 (Leikauf et al., 1981). Although transiently depressed following a single 60-minute exposure, the latter decreased clearance rates seen at 1000 μg/m^3 raise the possibility of more persistent or chronic depression of tracheobronchial clearance after repeated exposures to the same concentrations of H_2SO_4. The possible occurrence of such an effect in humans would be consistent with observations of persistently slowed clearance for several months following repeated exposures of donkeys to comparable H_2SO_4 concentrations (Schlesinger et al., 1978, 1979).

In studies with asthmatic subjects, no changes in airway function have been demonstrated after exposure to sulfuric acid and sulfate salts at concentrations less than 1000 μg/m^3. However, at higher concentrations (1000 μg/m^3), reduction in specific airway conductance (SG_{aw}) and forced expiratory volume ($FEV_{1.0}$) have been observed after sulfuric acid (H_2SO_4) and ammonium bisulfate (NH_4HSO_4) exposures as reported by Utell et al. (1981). No studies, on the other hand, have as yet evaluated the effects of sulfuric acid or other published sulfate salt aerosols on nasal or tracheobronchial mucus clearance functions.

13.6.3 Effects of Other Particulate Matter Species

Water-soluble sulfates have been the most frequent ingredients of experimental aerosol exposure atmospheres because ambient sulfate levels were earlier reported as likely being associated with morbidity epidemiologically. However, in addition to sulfuric acid and sulfates, other nonsulfur particulate matter species exist in the ambient air. These include POM, lead, arsenic, selenium, hydrogen ions, ammonium salts, and carbon as dust. Although controlled human exposure to some of these inherently toxic compounds is forbidden for obvious reasons, several investigators have conducted studies using carbon and other inert particles.

The relatively sparse results involving insoluble and other nonsulfur aerosols under controlled human exposure conditions preclude drawing conclusions regarding quantitative exposure/effect or dose/response relationships for the particulate chemical species studies. This is due to the fact that extremely high aerosol concentrations were typically employed in such studies. Nor can any clear conclusions be drawn, based on the available controlled human exposure data, in regard to size ranges of insoluble and other nonsulfur aerosols that may be associated with the induction of significant respiratory system effects at concentrations commonly found in the ambient air (although most of the controlled exposure studies generally appear to have employed either fine mode-sized particles < 2.5 μm diameter or inhalable particles < 10-15 μm diameter). However, the effects in polydispersed aerosol studies cannot be ascribed to fine particles alone. Only studies by McDermott (1962), Andersen et al. (1979), and Toyoma (1964) have clearly studied the effects of larger particles but at highly elevated levels of insoluble particulate matter not usually associated with ambient conditions.

13.7 REFERENCES

Abe, M. Effects of mixed nitrogen dioxide-sulfur dioxide on human pulmonary functions. Bull. Tokyo Med. Dent. Univ. 14:415-433, 1967.

Albert, R. E., J. R. Spiegelman, S. Shatsky, and M. Lippmann. The effect of acute exposure to cigarette smoke on bronchial clearance in the miniature donkey. Arch. Environ. Health 18:30-41, 1969.

Amdur, M. O. Toxicological appraisal of particulate matter, oxides of sulfur and sulfuric acid. J. Air Pollut. Control Assoc. 19:638-646, 1969.

Amdur, M. O. Animal studies. In: Proceedings of the Conference on Health Effects of Air Pollutants, National Academy of Sciences, Washington, DC, October 3-5, 1973. Serial No. 93-15, U.S. Senate, Committee on Public Works, Washington, DC, 1973. Washington, DC, 1973. pp. 175-205.

Amdur, M. O. The long road from Donora. 1974 Cummings Memorial Lecture. Am. Ind. Hyg. Assoc. J. 35:589-597, 1974.

Amdur, M. O., W. W. Melvin, Jr., and P. Drinker. Effects of inhalation of sulfur dioxide by man. Lancet 2:758-759, 1953.

Amdur, M. O., L. Silverman, and P. Drinker. Inhalation of sulfuric acid mist by human subjects. Arch. Ind. Hyg. Occup. Med. 6:305-313, 1952.

Andersen, I., L. Molhave, and D. F. Proctor. Human response to controlled levels of combinations of sulfur dioxide and inert dust. Scan. J. Environ. Health 7:1-7, 1981.

Andersen, I., G. R. Lundquist, P. L. Jensen, and D. F. Proctor. Human response to controlled levels of sulfur dioxide. Arch. Environ. Health 28:31-39, 1974.

Andersen, I., G. R. Lundquist, D. F. Proctor, and D. L. Swift. Human responses to controlled levels of inert dust. Am. Rev. Resp. Dis. 119-:619-627, 1979.

Andersen, I., P. L. Jensen, S. E. Reed, J. W. Craig, D. F. Proctor, and G. K. Adams. Induced rhinovirus infection under controlled exposure to sulfur dioxide. Arch. Environ. Health 32:120-126, 1977.

Arthur D. Little Incorporated. Research on Chemical Odors. Part I. Determination of Odor Thresholds for 53 Commercial Chemicals. The Manufacturing Chemists' Association, Washington, DC, January 1968.

Avol, E. L., M. P. Jones, R. M. Bailey, N. M-N. Chang, M. T. Kleinman, W. S. Linn, K. A. Bell, and J. D. Hackney. Controlled exposures of human volunteers to sulfate aerosols. Health effects and aerosol characterization. Am. Rev. Respir. Dis. 120:319-327, 1979.

Bates, D. V., and M. Hazucha. The short-term effects of ozone on the lung. In: Proceedings of the Conference on Health Effects of Air Pollutants, National Academy of Sciences, Washington, DC, October 35, 1973. Serial No. 93-15, U.S. Senate Committee on Public Works, Washington, DC, 1973. pp. 507-540.

Bedi, J. F., L. J. Folinsbee, S. M. Horvath, and R. S. Ebenstein. Human exposure to sulfur dioxide and ozone: absence of a synergistic effect. Arch. Environ. Health 34:233-239, 1979.

Bedi, J. F., S. M. Horvath, and L. J. Folinsbee. Human exposure to sulfur dioxide and ozone in a high temperature-humidity environment. Am. Ind. Hygi. Assoc. J., 1981 (in press).

Bell, K. A., and J. D. Hackney. Effects of Sulfate Aerosols upon Human Pulmonary Function. Coordinating Research Council, Inc. APRAC Project CA PM-27-75, 1977a.

Bell, K. A., W. S. Linn, M. Hazucha, J. D. Hackney, and D. V. Bates. Respiratory effects of exposure to ozone plus sulfur dioxide in Southern Californians and Eastern Canadians. Am. Ind. Hyg. Assoc. J. 38:696-706, 1977b.

Burton, G. G., M. Corn, J. B. L. Gee, D. Vassallo, and A. Thomas. Absence of "synergistic response" to inhaled low concentration gas-aerosol mixtures in healthy adult males. Presented at 9th Annual Air Pollution Medical Research Conference, Denver, Colorado, July 1968.

Burton, G. G., M. Corn, J. B. L. Gee, C. Vasallo, and A. P. Thomas. Response of healthy men to inhaled low concentrations of gas-aerosol mixtures. Arch. Environ. Health 18:681-692, 1969.

Bushtueva, K. A. The determination of the limit of allowable concentration of sulfuric acid in atmospheric air. In: Limits of Allowable Concentrations of Atmospheric Pollutants. Book 3. B. S. Levine, translator, U.S. Department of Commerce, Office of Technical Services, Washington, DC, 1957. pp. 20-36.

Bushtueva, K. A. Threshold reflex effect of SO_2 and sulfuric acid aerosol simultaneously present in the air. In: Limits of Allowable Concentrations of Atmospheric Pollutants. Book 4. B. S. Levine, translator, U.S. Department of Commerce, Office of Technical Services, Washington, DC, January 1961. pp. 72-79.

Bushtueva, K. A. New studies of the effect of sulfur dioxide and of sulfuric acid aerosol on reflex activity of man. In: Limits of Allowable Concentrations of Atmospheric Pollutants. Book 5. B. S. Levine, translator, U.S. Department of Commerce, Office of Technical Services, Washington, DC, March 1962. pp. 86-92.

Camner, P., K. Strandberg, and K. Philipson. Increased mucociliary transport by colinergic stimulation. Arch. Environ. Health 29:220-224, 1974.

Camner, P., K. Strandberg, and K. Philipson. Increased mucociliary transport by andenergic stimulation. Arch. Environ. Health 31:79-82, 1976.

Constantine, H., L. Dautrebande, N. Kaltreider, F. W. Lovejoy Jr., P. Morrow, and P. Perkins. Influence of carbachol, and of fine dust aerosols upon the breathing mechanics and the lung volumes of normal subjects and of patients with chronic respiratory disease before and after administering sympathominetic aerosols. Arch. Int. Pharmacodyn 123:239-251, 1959.

Corn, M. and R. T. Cheng. Interactions of sulfur dioxide with insoluble suspended particulate matter. J. Air Pollution Control Assoc. 22:870-875, 1972.

Cralley, L. V. The effect of irritant gases upon the rate of ciliary activity. J. Ind. Hyg. and Toxicol. 24:193-198, 1942.

D'Alfonso, D. A. The Limiting Factors of Nasal Respiration. Ph.D. Thesis, University of California, Santa Barbara, 1980.

Dodge, R. R., and B. Burrows. The prevalence and incidence of asthma and asthma-like symptoms in a general population sample. Am. Rev. Respir. Dis. 122:567-575, 1980.

Dubois, A. B., and L. Dautrebande. Acute effects of breathing inert dust particles and of carbachol aerosol on the mechanical characteristics of the lungs in man. Changes in response after inhaling sympathomimetic aerosols. J. of Clin. Invest. 37:1746-1754, 1958.

Dubrovskaya, F. I. Hygienic evaluation of pollution of atmospheric air of a large city with sulfur dioxide gas. In: Limits of Allowable Concentrations of Atmospheric Pollutants. Book 3. B. S. Levine, translator, U.S. Department of Commerce, Office of Technical Services, Washington, DC., 1957. pp. 37-51.

Fishchl, M. A., A. Pitchenik, and L. B. Gardner. An index predicting relapse and need for hospitalization in patients with acute bronchial asthma. N. Engl. J. Med. 305:783-789, 1981.

Frances, R., D. Alessandro, M. Lippmann, D. E. Proctor, and R. E. Albert. Effect of cigarette smoke on particle transport on nasociliary mucosa of donkeys. Arch. Environ. Health 21: 25-31, 1970.

Frank, N. R. Studies on the effects of acute exposure to sulfur dioxide in human subjects. Proc. R. Soc. Med. 57:1029-1033, 1964.

Frank, N. R., M. O. Amdur, and J. L. Whittenberger. A comparison of the acute effects of SO_2 administered alone or in combination with NaCl particles on the respiratory mechanics of healthy adults. Int. J. Air Water Pollut. 8:125-133, 1964.

Frank, N. R., M. O. Amdur, J. Worcester, and J. L. Whittenberger. Effects of acute controlled exposure to SO_2 on respiratory mechanics in healthy male adults. J. Appl. Physiol. 17:252-258, 1962.

Frank, R., C. E. McJilton, and R. J. Charlson. Sulfur oxides and particles; effects on pulmonary physiology in man and animals. In: Proceedings of Conference on Health Effects of Air Pollution. National Academy of Sciences, Washington, DC, October 3-5, 1973. Serial No. 93-15, U.S. Senate, Committee on Public Works, Washington, DC, 1973. pp. 207-225.

Franklin, W. Asthma in the emergency room. Assessment and treatment. N. Engl. J. Med. 305:826-827, 1981.

Gökenmeijer, J. D. M., K. DeVries, and N. G. M. Orie. Response of the bronchial tree to chemical stimuli. Rev. Inst. Hyg. Mines (Hasselt) 28:195-197, 1973.

Greenwald, I. Effects of inhalation of low concentrations of sulfur dioxide upon man and other mammals. Arch. Ind. Hyg. Occup. Med. 10:455-475, 1954.

Gunnison, A. F., and E. D. Palmes. S-Sulfonates in human plasma following inhalation of sulfur dioxide. Am. Ind. Hyg. Assoc. J. 35:288-291, 1974.

Hazucha, M., and D. V. Bates. Combined effect of ozone and sulphur dioxide on human pulmonary function. Nature (London) 257:50-51, 1975.

Holmes, J. A., E. C. Franklin, and R. A. Gould. Report of the Selby Smelter Commission. Bureau of Mines Bulletin 98, U.S. Department of the Interior, Washington, DC, 1915.

Horvath, S. M., and L. J. Folinsbee. Interactions of Two Air Pollutants, Sulfur Dioxide and Ozone, on Lung Functions. Grant ARB-4-1266, California Air Resources Board, Sacramento, CA, March 1977.

Horvath, S. M., L. J. Folinsbee, and J. F. Bedi. Effects of large (0.9 μm) sulfuric acid aerosols on human pulmonary function. Environ. Res., 1981 (in press).

Islam, M. S., and W. T. Ulmer. The effects of long-time exposure (8 h per day on 4 successive days) to a gas mixture of SO_2 + NO_2 + and O_3 in the threefold MIC range (maximum emission concentration) on lung function and reactivity of the bronchial system of healthy persons. Wissenschaft und Unwelt 4:186-190, 1979(b).

Islam, M. S., and W. T. Ulmer. The influence of acute exposure against a combination of 5.0 ppm SO_2, 5.0 ppm NO_2, and 0.1 ppm O_3 on the lung function in the MAK (lower toxic limit) area (short-time test). Wissenschaft und Unwelt 3:131-137, 1979(a).

Jaeger, M. J., D. Tribble, and H. J. Wittig. Effect of 0.5 ppm sulfur dioxide on the respiratory function of normal and asthmatic subjects. Lung 156:119-127, 1979.

Kagawa, J., and K. Tsuru. Respiratory effect of 2-hour exposure with intermittent exercise to ozone and sulfur dioxide alone and in combination in normal subjects. Jap. J. Hyg. 34:690-696, 1979.

Kerr, H. D., T. J. Kulle, B. P. Farrell, L. R. Sauder, J. L. Young, D. L. Swift, and R. M. Borushok. Effects of sulfuric acid aerosol on pulmonary function in human subjects. Environmental Research, 26:42-50, 1981.

Kisskalt, K. Über den Einfluss der inhalation schwelfiger Säure auf die Entevickelung der Lungentuberculose: Ein Bietrag zum Studien der Gewerbekrankheiten. [Effects of inhaling sulfuric acid on the development of tuberculoses of the lungs. A continuation to studies of occupational diseases.] Z. Hyg. 48:269-279, 1904.

Kleinman, M. T., and J. D. Hackney. Effects of sulfate aerosols upon human pulmonary function. APRAE Project CAPM-27-75, Coordinating Research Council, Inc., New York, NY, 1978.

Kleinman, M. T., R. M. Bailey, Y. C. Chang, K. W. Clark, M. P. Jones, W. S. Linn, and J. D. Hackney. Exposures of human volunteers to a controlled atmospheric mixture of ozone, sulfur dioxide and sulfuric acid. Am. Indus. Hyg. Assoc. J. 42:61-69, 1981.

Koenig, J. Q., W. E. Pierson, and R. Frank. Acute effects of inhaled SO_2 plus NaCl droplet aerosol on pulmonary function in asthmatic adolescents. Environ. Res. 22:145-153, 1980.

Koenig, J. Q., W. E. Pierson, M. Horike, and R. Frank. Effects of SO_2 plus NaCl aerosol combined with moderate exercise on pulmonary function in asthmatic adolescents. Environ. Res. 25:340-348, 1981.

Kreisman, H., C. A. Mitchell, H. R. Hosein, and A. Bouhuys. Effect of low concentrations of sulfur dioxide on respiratory function in man. Lung 154:25-34, 1976.

Lawther, P. J. Effects of inhalation of sulfur dioxide on respiration and pulse rates in normal subjects. Lancet 2:745-748, 1955.

Lawther, P. J., A. J. MacFarlane, R. E. Waller, and A. G. F. Brooks. Pulmonary function and sulphur dioxide, some preliminary findings. Environ. Res. 10:355-367, 1975.

Lehmann, K. B. Experimentelle Studien über den Einfluss technisch und hygienisch wichtiger Gase und Dämpfe auf den Organismus. VI. Schwefliger Säure. [Experimental studies on the effects on the system of technically and hygienically important gases and fumes. VI. Sulfuric Acid.] Arch. Hyg. 18:180-191, 1893.

Leikauf, G., D. B. Yeates, K. A. Wales, D. Spedtor, R. E. Albert, and M. Lippmann. Effects of sulfuric acid aerosol on respiratory mechanics and mucociliary particle clearance in healthy nonsmoking adults. Am. Ind. Hyg. Assoc. J. 42:273-282, 1981.

Linn, W. S., R. M. Bailey, D. A. Medway, J. G. Venet, L. H. Wightman, and J. D. Hackney. Respiratory responses of young adult asthmatics to sulfur dioxide exposure under simulated ambient conditions. Environ. Res. 1982 (in press).

Linn, W. S., M. P. Jones, R. M. Bailey, M. T. Kleinman, C. E. Spier, R. D. Fischer, and J. D. Hackney. Respiratory effects of mixed nitrogen dioxide and sulfur dioxide in human volunteers under simulated ambient exposure conditions. Environ. Res. 22:431-438, 1980.

Lippman, M., R. E. Albert, D. B. Yeats, K. Wales, and G. LeiKauf. Effect of sulfuric acid mist on mucociliary bronchial clearance in healthy non-smoking humans. In: Aerosols in Science, Medicine and Technology - The Biomedical Influence of the Aerosol, Association for Aerosol Research, Düsseldorf, German (FRG), October 3-5, 1979. W. Stöber and R. Jaenicke, eds., Association for Aerosol Research, Mainz, Germany (FRG), 1980. pp. 157-162.

McDermott, M. Acute respiratory effects of the inhalation of coal-dust particles. J. Physiol. 162:53 p. 1962.

McJilton, C. E., R. Frank, and R. J. Charlson. Influence of relative humidity on functional effects of an inhaled SO_2-aerosol mixture. Am. Rev. Respir. Dis. 113:163-169, 1976.

McKerrow, C. B. Chronic respiratory disease in Great Britain. Arch. Environ. Health 8: 182-187, 1964.

Melville, G. N. Changes in specific airway conductance in healthy volunteers following nasal and oral inhalation of SO_2. West Indian Med. J. 19:231-235, 1970.

Nadel, J., H. Salem, B. Tamplin, and Y. Tokiwa. Mechanism of bronchoconstriction during inhalation of sulfur dioxide. J. Appl. Physiol. 20:164-167, 1965.

Nakamura, K. Response of pulmonary airway resistance by interaction of aerosols and gases in different physical and chemical nature. Nippon Eiseigaku Zasshi 19:38-50, 1964.

Newhouse, M. T., M. Dolovich, G. Obminski, and R. K. Wolff. Effect of TLV levels of SO_2 and H_2SO_4 on bronchial clearance in exercising man. Arch. Environ. Health 33:24-32, 1978.

Niinimaa, V., P. Cole, S. Mintz, and R. J. Shephard. Oronasal distribution of respiratory flow. Resp. Physiol. 43:69-75, 1981.

Niinimaa, V., P. Cole, S. Mintz, and R. J. Shephard. The switching point from nasal to oronasal breathing. Resp. Physiol. 42:61-71, 1980.

Norris, R. M., and J. M. Bishop. The effect of calcium carbonate dust on ventilation and respiratory gas exchange in normal subjects and in patients with asthma and chronic bronchitis. Clin. Sci. 30:103-115, 1966.

Ogata, M. Über die Giftigkeit der schweffigen Säure. [The Toxicity of Sulfuric Acid.] Arch. Hyg. 2:223-245, 1884.

Reichel, G. The effect of sulfur dioxide on the airway resistance of man. Annual Meeting of the German Scoiety for Industrial Medicine, 1972.

Ryazanov, V. A. Sensory physiology as basis for air quality standards. Arch. Environ. Health 5:479-494, 1962.

Sackner, M. A., D. Ford, R. Fernandez, J. Cipley, D. Perez, M. Kwocka, M. Reinhart, E. D. Michaelson, R. Schreck, and A. Wanner. Effects of sulfuric acid aerosol on cardiopulmonary function of dogs, sheep and humans. Am. Rev. Respir. Dis. 118:497-510, 1978.

Saibene, F., P. Magnoni, C. L. Lafortuna, and R. Mostardi. Oronasal breathing during exercise. Pfluegers Arch. 378:65-69, 1978.

Schlesinger, R. B., M. Lippmann, and R. E. Albert. Effects of short-term exposures to sulfuric acid and ammonium sulfate aerosols upon bronchial airway function in the donkey. Am. Ind. Hyg. Assoc. J. 39:275-286, 1978.

Schlesinger, R. B., M. Halpern, R. E. Albert, and M. Lippmann. Effect of chronic inhalation of sulfuric acid mist upon mucociliary clearance from the lungs of donkeys. J. Environ. Pathol. Toxicol. 2:1351-1367, 1979.

Shalamberidze, O. P. Reflex effects of mixtures of sulfur and nitrogen dioxides. Hyg. Sanit. 32:10-15, 1967.

Sheppard, D., A. Saisho, J. A. Nadel, and H. A. Boushey. Exercise increases sulfur dioxide-induced bronchoconstriction in asthmatic subjects. Am. Rev. Respir. Dis. 123:486-491, 1981.

Sheppard, D., W. S. Wong, C. F. Uehara, J. A. Nadel, and H. A. Boushey. Lower threshold and greater bronchomotor responsiveness of asthmatic subjects to sulfur dioxide. Am Rev. Respir. Dis. 122:873-878, 1980.

Sim, V. M., and R. E. Pattle. Effect of possible smog irritants on human subjects. J. Am. Med. Assoc. 165:1908-1913, 1957.

Snell, R. E., and P. C. Luchsinger. Effects of sulfur dioxide on expiratory flow rates and total respiratory resistance in normal human subjects. Arch. Environ. Health 18:693-698, 1969.

Speizer, F. E., and N. R. Frank. A comparison of changes in pulmonary flow resistance in health volunteers acutely exposed to SO_2 by mouth and by nose. Br. J. Ind. Med. 23:75-79, 1966a.

Speizer, F. E., and N. R. Frank. The uptake and release of SO_2 by the human nose. Arch. Environ. Health 12:725-728, 1966b.

Stacy, R. W., D. E. House, M. Friedman, M. Hazucha, J. Green, L. Raggio, and L. J. Roger. Effects of 0.75 ppm sulfur dioxide on pulmonary function parameters of normal human subjects. Arch. Environ. Health, 36:172-178, 1981.

Stahlhofen, W., J. Gebhart, and J. Heyden. Experimental determination of the regional deposition of aerosol particles in the human respiratory tract. Am. Ind. Hyg. Assoc. J. 41:385-398, 1980.

Tomono, Y. Effects of SO_2 on human pulmonary functions. Sangyo Igaku 3:77-85, 1961.

Toyama, T. Studies on aerosols. Synergistic response of the pulmonary airway resistance of inhaling sodium chloride aerosols and SO_2 in man. Sangyo Igaku 4:86-92, 1962.

Toyama, T. Air pollution and its helath effects in Japan. Arch. Environ. Health 8:153-173, 1964.

Toyama, T., and K. Nakamura. Synergistic response to hydrogen perixide aerosols and sulfur dioxide to pulmonary airway resistance. Ind. Health 2:34-45, 1964.

United States Department of Health, Education and Welfare. Prevalence of selected chronic respiratory conditions in the United States - 1970. Publication No. (HRA) 74-1511, Sept. 1973 by Center for Health Statistics, Rockville, MD.

Ulmer, W. T. Inhalative noxen: schwefeldioxyd. Toxic Inhalation: Sulfur Dioxide. Pneumonologie 150:83-96, 1974.

Utell, M. J., A. T. Aquilina, W. J. Hall, D. M. Speers, R. G. Douglas, Jr., F. R. Gibb, P. E. Morrow, and R. W. White. Development of airway reactivity to nitrates in subjects with influenza. Am. Rev. Respir. Dis. 121:233-241, 1980.

Utell, M. J., P. E. Morrow, and R. W. Hyde. Inhaled Particles. V. Proceedings of the 5th International Symposium, Cardiff, Wales, September 1980, Pergamon Press, London, England, 1981 (in press).

von Nieding, G., H. M. Wagner, H. Krekeler, H. Löllgen, W. Fries, and A. Beuthan. Controlled studies of human exposure to single and combined action of NO_2, O_3 and SO_2. Int. Arch. Occup. Environ. Health 43:195-210, 1979.

Weir, F. W., and P. A. Bromberg. Further investigation of the effects of sulfur dioxide on human subjects. Annual Report Project No. CAWC S-15, American Petroleum Institute, Washington, DC, 1972. pp. 1-74.

Weir, F. W., and P. A. Bromberg. Effects of sulfur dioxide on human subjects exhibiting peripheral airway impairment. Project No. CAWC S-15, American Petroleum Institute, September 1973. pp. 1-40.

Widdicombe, J. G., D. C. Kent, and J. A. Nadel. Mechanism of bronchoconstriction during inhalation of dust. J. Appl. Physidogy 17:613-616, 1962.

Wolff, R. K., M. Dolovich, C. M. Rossman, and M. T. Newhouse. Sulphur dioxide and tracheo-bronchial clearance in man. Arch. Environ. Health 30:521-527, 1975a.

Wolff, R. K., M. Dolovich, G. Obminski, and M. T. Newhouse. In: Effect of sulfur dioxide on trachiobronchial clearance at rest and during exercise. Inhaled Particles, Proceedings of the 4th International Symposium, Edinburgh, Scotland, September 22-26, 1975. W. H. Walton, ed., Pergamon Press, London, England, 1975b. pp. 321-332.

Yamada, J. Untersuchunger über die quantitative Absorption der Dämpfe einiger Säuren durch Tier und Mensch. [Investigations on the quantitative absorption of some acid fumes in animals and man.] Ph.D. Thesis, Würzburg, 1905. (See Lehmann, K. B., Arch. Hyg. 67:57-98, 1908.)

APPENDIX 13A

The following is an abstract of a manuscript recently submitted for peer review and publication, but not yet available in the published literature. This abstract is included here strictly for informational purposes and cannot presently be definitely analyzed for possible consideration as part of this criteria document.*

EFFECT OF THE ORONASAL BREATHING ROUTE ON THE BRONCHOCONSTRICTOR RESPONSE TO SULFUR DIOXIDE IN EXERCISING ASTHMATIC SUBJECTS. M. B. Kirkpatrick, D. Sheppard, J. A. Nadel, H. A. Boushey.

Cardiovascular Research Institute, University of California, San Francisco.

We studied how the oronasal breathing route during exercise affects the bronchoconstrictor response to inhaled sulfur dioxide (SO_3) in asthmatic subjects. In six subjects, we compared the changes in specific airway resistance (SR_{aw}) caused by breathing humidified air through a mouthpiece during 5 min of exercise on a bicycle ergometer (550 kpm/min) to the changes caused by breathing humidified air plus 0.5 ppm of SO_2 by mouthpiece, by facemask, and by facemask with the mouth occluded (nose breathing) during exercise. Breathing humidified air plus 0.5 ppm SO_2 by both mouthpiece and facemask significantly increased SR_{aw} in all 6 subjects; breathing SO_2 by nose significantly increased SR_{aw} in 5 of 6 subjects. Although the increase in SR_{aw} caused by breathing SO_2 varied considerably among subjects, for the group, breathing SO_2 by all 3 routes increased SR_{aw} (mouthpiece, from 6.8 ± 4.0 to 16.4 ± 9.0 L x cm H_2O/L/S [mean ± S. D.], facemask, from 7.4 ± 3.6 to 12.4 ± 5.9, nose only, 6.4 ± 2.7 to 10.6 ± 5.2) significantly more than breathing humidified air without SO_2 through a mouthpiece (from 7.2 ± 6.2 to 8.3 ± 6.8) ($p < 0.05$ for each route of breathing SO_2 compared to breathing air without SO_2). Breathing SO_2 through a mouthpiece increased SR_{aw} significantly more than breathing SO_2 by facemask with the mouth occluded ($p < 0.05$) but not significanlty more than breathing SO_2 by facemask ($p < 0.05$). These results indicate that, although nasal breathing partially protects against SO_2-induced bronchoconstriction, both oral and oronasal breathing low concentrations of SO_2 during exercise can cause significant bronchoconstriction in people with asthma.

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*Note: Since finalization of the present criteria document in December 1981, this and certain other controlled human exposure studies of SO_2 effects on asthmatics appeared in the published literature and were then later evaluated in an addendum to the present document. That addendum is included in Volume I after Chapter 1 (Executive Summary and Conclusions) of this document.

14. EPIDEMIOLOGICAL STUDIES ON THE EFFECTS OF PARTICULATE MATTER AND SULFUR OXIDES ON HUMAN HEALTH

14.1 INTRODUCTION

This chapter evaluates epidemiological literature concerning health effects associated with ambient air exposures to sulfur oxides and particulate matter. The main focus of the chapter is on: (1) qualitative characterization of human health effects associated with exposure to airborne sulfur dioxide (SO_2), related sulfur compounds, and other particulate matter (PM); (2) quantitative delineation of exposure-effect and exposure-response relationships for induction of such effects; and (3) identification of population groups at special risk for experiencing the effects at ambient exposure levels.

The epidemiological data discussed here both complement and extend information presented as part of health effects analyses in preceding chapters (11,12,13) of this document. Those chapters focus on information from animal toxicology and controlled human exposure studies, which offer the advantage of characterizing, under well-controlled laboratory conditions, differential patterns of respiratory tract deposition and clearance of: SO_2; sulfates (SO_4) and sulfuric acid (H_2SO_4); and other particulate matter of varying size and chemical composition. Also, the animal toxicological studies provide evidence for notable health effects occurring in mammalian species as the result of respiratory tract deposition of SO_2 and PM, including: transient alterations in pulmonary functions; altered mucociliary clearance and other respiratory tract defense mechanisms; and increased susceptibility to infection and morphological damage seen especially after high-level or prolonged exposures. However, while such results from animal studies are highly suggestive of analogous effects possibly being induced in human beings, caution must be exercised in directly extrapolating the findings or associated dose-effect relationships to human health. More direct delineation of quantitative dose-effect or dose-response relationships is possible through controlled human exposure studies, but such studies also have important limitations. For example, whereas controlled human exposure studies have demonstrated SO_2 or PM induction of transient pulmonary function decrements, altered mucociliary clearance patterns, and symptomatic effects consistent with animal toxicology study findings, observation of such effects has generally been confined to conditions involving single or a few repeated short-term (<3 hours) exposures, rather than prolonged chronic exposure conditions. Also left unanswered by controlled human exposure studies are questions concerning whether or not more severe effects (e.g., increased vulnerability to respiratory diseases or marked morphological damage) are associated with either short-term or prolonged ambient exposure conditions.

Epidemiological studies, in contrast, offer several advantages beyond those of animal toxicology or controlled human exposure studies. Health effects of both short- and long-term pollutant exposures (including complex mixtures of pollutants) can be studied and sensitive members of populations at special risk for particular effects at ambient air concentrations identified. Also, epidemiological evaluations allow for investigation of both acute and chronic disease effects and associated human mortality. Epidemiological studies, then, together with the results of controlled animal and human exposure studies, can significantly contribute to more complete understanding of health effects of sulfur oxides and particulate matter, especially in helping to characterize human health effects associated with those pollutants under ambient conditions. Despite such advantages, however, important limitations do exist with regard to the conduct, analysis, interpretation, and use of many of the available epidemiological studies on the health impact of SO_2 and PM, as discussed next.

14.1.1 Methodological Considerations

As noted by Lowrence (1976), epidemiological and other types of studies employed in generating information relevant to human risk assessment typically focus on one or more of four areas of investigation: (1) defining exposure conditions; (2) identifying adverse effects; (3) relating exposures to effects; and (4) estimating overall risk of specific population groups experiencing particular effects under various exposure conditions.

In relation to accomplishing these goals, one important limitation of most epidemiological studies reviewed here has been less-than-optimum characterization of community air quality parameters used to estimate exposures of population groups to varying atmospheric concentrations of SO_2 and PM. Such characterization of air quality has generally involved relatively crude estimates of levels of pollutants present, often allowing for only limited qualitative statements to be made regarding exposure conditions--e.g., whether a given site or time period had comparatively higher or lower atmospheric levels of SO_2 or PM than some other site or time period. Only rarely have the epidemiological studies relied on measurement methods or their practical field applications that permit reasonably precise determinations of variations in ambient levels of the pollutants so as to adequately quantitate SO_2 or PM levels associated with observed health effects. Even when reasonable quantification of community air quality parameters was achieved, however, the use of such data in estimating actual population exposures has typically been further constrained by factors such as siting of air sampling devices in relation to study populations, frequency and duration of sampling periods, activity patterns of study population members, and contributions of indoor air pollution to overall exposures of study groups. These, and other limitations noted, arise in part from the fact that most of the presently-reviewed epidemiological studies utilized air quality monitoring data obtained from sampling networks originally established for purposes other than health-related research and, therefore, not optimally designed to provide the specific types or quality of aerometric data needed for epidemiological assessment of health effects related to SO_2 and PM. Therefore, the aerometric data reported should generally be viewed as yielding, at best, only approximate estimates of actual study population exposures.

Adequate characterization of health effects associated with various SO_2 and PM exposure conditions has represented a second major problem for most of the epidemiological studies evaluated here. Various health endpoint measurements (mortality, morbidity, and indirect measures of morbidity) have been employed in such studies and each has advantages and disadvantages, as discussed elsewhere (Hill, 1965; Speizer, 1969; Holland, 1970; Higgins, 1974; Goldsmith and Friberg, 1977; American Thoracic Society, 1978). Some health outcome measurements have involved direct observations of signs and symptoms of disease states or objective indicators thought to be associated with the occurrence of illnesses, e.g., patient visits to hospitals or clinics or absenteeism from school or work. Direct quantification of health effects has also included measurement of biochemical or physiological changes in study populations, as in recording of pulmonary function changes by spirometry methods. Indirect measures or indices of health effects have also been used, e.g., in gathering information on frequency and duration of respiratory illnesses by telephone interviews, written questionnaires, or self-reported entries in diaries. The validity of such indirect measurements of health effects, however, is highly dependent on the ability and motivation of respondents to recall and report accurately past or present health-related events; this can be influenced by numerous extraneous factors such as age, cultural and educational background, instructions from experimenters, sequencing of questions, and interviewer variability or bias. Confidence in results obtained by either direct or indirect measurement methods is enhanced if potential interfering or biasing factors have been appropriately controlled for and, especially for indirect health endpoint measurements, if results have been validated against corroborating evidence.

Adequately relating observed health effects to specific parameters of ambient exposure conditions is another objective not often achieved by epidemiological studies reviewed below, such that few allow for confident qualitative or quantitative characterization of SO_2 or PM exposure-health effect relationships. For example, competing risks, such as cigarette smoking and occupational exposures, may contribute to observed health effects results and, therefore, usually must be controlled for or taken into account in order for much confidence to be placed in reported air pollution-health effects relationships; however, many studies on SO_2 or PM effects have not adequately controlled for such factors. Similarly, possible effects of other covarying or confounding factors such as socioeconomic status, race, and meteorological parameters have not always been adequately controlled for or evaluated. Also, further complicating the evaluation of the epidemiological data is the fact that exposure parameters are not subject to experimenter control; thus, ambient levels of a given pollutant often vary widely over the course of a study. This makes it extremely difficult to determine whether mean concentrations, peak concentrations, rapid fluctuations in levels, or other air quality factors were most important as determinants of reported health effects. Significant covariation among concentrations of SO_2, PM, and other pollutants has also often made it very difficult to distinguish among their relative contributions to observed health effects.

Estimation of overall risk by means of epidemiological studies requires still further steps beyond delineation of exposure-effect relationships that define exposure conditions (levels, durations, etc.) associated with induction of specific health effects. That is, risk estimation also requires: (1) identification of particular population groups likely to manifest health effects under exposure conditions of concern; and (2) ideally, determination of numbers or percentages of such individuals (responders) likely to be affected at various exposure or dose levels. Delineation of the former, i.e., identification of population groups at special risk at lower exposure levels of SO_2 and PM than other groups, has only started to be accomplished via the epidemiological studies reviewed here. Also, epidemiological delineation of quantitative dose-response (or, more correctly, exposure-response) relationships, defining percentages of population groups likely to manifest a given health effect at various levels or durations of exposure to SO_2 and PM, is largely lacking at this time.

Another limitation of epidemiological information reviewed here concerns its usefulness in demonstrating cause-effect relationships versus merely establishing associations (which may be non-causal in nature) between various health effects and SO_2 or PM. The interpretation of epidemiological data as an aid in inferring causal relationships between presumed causal agents and associated effects has been previously discussed by several expert committees or deliberative bodies faced with evaluation of controversial biomedical issues (U.S. Surgeon General's Advisory Committee on Smoking and Health, 1964; U.S. Senate Committee on Public Works, Subcommittee on Air and Water Pollution, 1968). Among criteria selected by each group for determination of causality, the following were included: (1) the strength of the association; (2) the consistency of the association, as evidenced by its repeated observation by different persons, in different places, circumstances and time; (3) specificity of the association; (4) the temporal relationship of the association; (5) the coherence of the association in being consistent with other known facts; (6) the existence of a biological gradient, or dose-reponse curve, as revealed by the association; and (7) the biological plausibility of the association. In discussing the use of such criteria, Hill (1965) further noted that strong support for likely causality suggested by an association may be derived from experimental or semi-experimental evidence, where manipulation of the presumed causative agent (its presence or absence, variability in intensity, etc.) also affects the frequency or intensity of the associated effects. Importantly, both Hill (1965) and the deliberative bodies or expert committees were careful to emphasize, regardless of the specific set of criteria selected by each, that no one criterion is definitive by itself nor is it necessary that all be fulfilled in order to support a determination of causality. Also, Hill (1965) and several of the expert groups noted that statistical methods cannot establish proof of a causal relationship in an association nor does lack of "statistical significance" of an association according to arbitrarily selected probability criteria necessarily negate the possibility of a causal relationship. That is, as stated by the U.S. Surgeon General's Advisory Committee on Smoking and Health (1964): "The causal significance of an association is a matter of judgment which goes beyond any statement of statistical probability."

14.1.2 Guidelines for Assessment of Epidemiological Studies

Taking into account the above methodological limitations, the following set of guidelines can be stated by which to judge the relative scientific quality of epidemiological studies and their findings reviewed here:

1. Was the quality of the aerometric data used sufficient to allow for meaningful characterization of geographic or temporal differences in study population pollutant exposures in the range(s) of pollutant concentrations evaluated?
2. Were the study populations well defined and adequately selected so as to allow for meaningful comparisons between study groups or meaningful temporal analyses of health effects results?
3. Were the health endpoint measurements meaningful and reliable, including clear definition of diagnostic criteria utilized and consistency in obtaining dependent variable measurements?
4. Were the statistical analyses employed appropriate and properly performed and interpreted, including accurate data handling and transfer during analyses?
5. Were potentially confounding or covarying factors adequately controlled for or taken into account in the study design and statistical analyses?
6. Are the reported findings internally consistent, biologically plausible, and coherent in terms of consistency with other known facts?

Few, if any, epidemiological studies deal with all of the above points in a completely ideal fashion; nevertheless, these guidelines provide benchmarks for judging the relative quality of various studies and for selecting the best for detailed discussion here. Detailed critical analysis of all epidemiological studies on health effects of SO_2 and PM, especially in relation to all of the above questions, represents an undertaking beyond the scope of the present document. Of most importance for present purposes are those studies which provide useful quantitative information on exposure-effect or exposure-response relationships for health effects associated with ambient air levels of SO_2 and PM likely to be encountered in the United States during the next 5 years. Accordingly, the following criteria were employed in selecting studies for detailed discussion in the ensuing text:

1. Concentrations of both SO_2 and PM were reported, allowing for potential evaluation of their separate or combined effects.
2. Study results provide information on quantitative relationships between health effects and ambient air SO_2 and PM levels of current concern (i.e., generally $\leq$ 1000 $\mu g/m^3$).
3. Important methodological considerations were adequately addressed, especially (a) in controlling for likely potentially confounding factors and (b) in carrying out data collection, analysis, and interpretation so as to minimize errors or potential biases which could be reasonably expected to affect the results.
4. The study results have been reported in the open literature or are in press, typically after having undergone peer review.

In addition, some studies not meeting all of the above criteria are briefly mentioned or discussed in the main text below as appropriate to help elucidate particular points concerning the health effects of SO_2 and/or PM. Additional studies, evaluated by the present authors but found to be of very limited usefulness for present criteria development purposes are noted in Appendix 14A, along with annotated comments on specific methodological aspects associated with each that limit their results to qualitative findings only or make clear attribution of reported health effects to SO_2 or PM questionable based on their reported results.

As a starting point in this assessment, key information from Chapters 2 and 3 is summarized regarding physical and chemical properties of SO_2 and PM indexed by air quality measurements used in epidemiological studies evaluated in this chapter. The ensuing discussion of community health epidemiological studies is then subdivided into two subsections: Section 14.3 deals with studies of acute mortality and morbidity effects most germane to development of health criteria for possible short-term (e.g., 24 hour) ambient air standards; and Section 14.4 discusses studies of mortality and morbidity effects associated with chronic exposures most pertinent for development of health criteria for long-term (annual average) ambient air standards. The last section (14.5) provides an integrative summary and interpretation of the overall pattern of results evaluated in preceding sections.

The extensive epidemiological literature on the effects of occupational exposures to SO_2 and PM presently available is not reviewed here for several reasons:

1. Such literature generally deals with effects of exposures to SO_2 or PM chemical species at levels many times higher than those encountered in the ambient air by the general population.

2. Populations exposed occupationally mainly include healthy adults, self-selected to some extent in terms of being better able to tolerate exposures to SO_2 or PM substances than more susceptible workers seeking alternative employment or other groups often at special risk among the general public (e.g., the old, the chronically ill, young children, and asthmatics).

3. Extrapolation of observed occupational exposure-health effects relationships (or lack thereof) to the general public (especially population groups at special risk) could, therefore, be potentially misleading in terms of demonstrating health effects among healthy workers at higher exposure levels than would affect susceptible groups in the general population.

The occupational literature does, however, demonstrate links between acute high level or chronic lower level exposures to SO_2 or many different PM chemical species and a variety of health effects, including: pulmonary function changes; respiratory tract diseases; morphological damage to the respiratory system; and respiratory tract cancers. The reader is referred to National Institute of Occupational Safety and Health (NIOSH) criteria documents and other assessments listed in Appendix 14B for information on health effects associated with occupational airborne exposures to SO_2 and various PM species.

14.2 AIR QUALITY MEASUREMENTS

Of key importance for evaluation of epidemiological studies reviewed here is a clear understanding of the limitations of the analytical methods employed in determining ambient air aerometric data (PM and SO_2 levels) utilized in those studies. These methods are discussed in more detail in Chapter 3.

14.2.1 Sulfur Oxides Measurements

Three main measurement methods or variations thereof were used to generate data cited for sulfur dioxide (SO_2) levels in epidemiological studies discussed below: (1) sulfation rate (lead dioxide); (2) hydrogen peroxide; and (3) the West-Gaeke (pararosaniline) methods. With sulfation rate methods, airborne sulfur compounds react with lead dioxide in a paste spread over an atmospherically exposed plate or cylinder. The sulfur compound reaction rate is expressed in $SO_3/cm^2/day$; but the reaction is not SO_2-specific, and atmospheric concentrations of SO_2 or other sulfur compounds cannot be accurately extrapolated from the results, which are markedly affected by variations in temperature and humidity. Lead dioxide gauges were widely used in the United Kingdom prior to 1960 and provided aerometric data reported in some British epidemiological studies; sulfation rate methods were also used in some American studies.

Use of the hydrogen peroxide method was gradually expanded in the United Kingdom during the 1950s (usually in tandem with apparatus for PM (smoke) monitoring) and the method was adopted in the early 1960s as the standard SO_2 method used for the United Kingdom National Survey of Air Pollution and, as an OECD-recommended method, elsewhere in Europe. The method can yield reasonably accurate estimates of atmospheric SO_2 concentrations expressed in $\mu g/m^3$; but the results can be affected by factors such as temperature, presence of atmospheric ammonia and titration errors. Little quality assurance information exists on sources and magnitudes of errors encountered in the use of the method to obtain SO_2 data reported in specific British or European epidemiological studies, making it difficult to assess the accuracy or precision of reported SO_2 values. Even the extensive quality assurance information reported (Warren Spring Laboratory, 1961; 1962; 1966; 1967; 1975; 1977; OECD, 1965; Ellison, 1968) for SO_2 measurements made by the method for the United Kingdom National Survey is of very limited use in evaluating the quality of specific SO_2 results reported in various British epidemiological studies.

The West-Gaeke (pararosaniline) method more widely used in the United States involves absorption of SO_2 in potassium tetrachloromercurate solution, producing a chemical complex reacted with pararosaniline to form a red-purple color measured colorimetrically. The method, suitable for sampling up to 24 hours, is specific for SO_2 if properly implemented to minimize interference by nitrogen or metal oxides, but results can be affected by factors such as temperature variations and mishandling of reagents. Limited quality assurance information (U.S. Congress, House of Representatives, 1976) has been reported for some American SO_2 measurements by the West-Gaeke method but is generally lacking by which to evaluate the SO_2 data reported in most published American epidemiological studies.

Measurement approaches for suspended sulfates and sulfuric acid, used mainly in the United States, include turbidimetric and methylthymol blue methods. The former usually involves collection of samples on sulfate-free glass fiber filters by high-volume PM samplers. Sulfate is extracted and precipitated with barium chloride, and turbidity of the suspension is determined spectrophotometrically or nephelometrically. However, the method does not differentiate between sulfates and sulfuric acid, and secondary formation of such products from SO_2 in air drawn through the filter can affect estimation of atmospheric sulfate levels. Similar collection procedures and limitations apply for the methylthymol blue method. Thus, these two methods are not specific measures of suspended sulfates, and their results can only serve as rough indicators of atmospheric levels of SO_x-related PM.

14.2.2 Particulate Matter Measurements

To be of maximum value, epidemiological studies on PM effects must utilize aerometric methods that provide meaningful data, not only regarding the mass or amount of atmospheric PM but also quantitative information on the size and chemical composition of particles present. In actual practice, most epidemiological studies on PM effects have relied on air quality data from air monitoring instruments of questionable sampling accuracy and not specifically designed for health-related research. The resulting data have thus typically only provided limited information regarding mass, size or chemical properties of the PM sampled.

Three measurement approaches or variations were mainly used to obtain PM data cited in epidemiological studies reviewed below: (1) the British Smoke light reflectance method or variations used in the United Kingdom and elsewhere in Europe; (2) the American Society for Testing and Materials (ASTM) filter-soiling light transmittance method used in the United States; and (3) the high-volume sampling method widely employed in the United States.

The British Smoke (BS) method and its variations in routine use typically employed standard monitoring equipment with a D_{50} cutpoint of $\approx$ 4.5 μm (McFarland et al., 1982). Thus, whether or not larger atmospheric coarse-mode particles were present during the sampling period, predominantly small particles were collected. The D_{50} of the instrument may, however, shift at higher windspeeds. The BS method neither directly measures the mass nor determines chemical composition of collected particles. Rather, it primarily measures reflectance of light from a stain formed by particles collected on filter paper somewhat inefficient for collecting very fine particles (Liu et al., 1978). Reflectance of light from the stain depends both on density of the stain or amount of PM collected in a standard period of time and optical properties of the collected materials. Smoke particles composed of elemental carbon of the type found in incomplete fossil fuel combustion products typically make the greatest contribution to stain darkness, especially in urban areas. Thus, the amount of elemental carbon, but not organic carbon, present in the stain is most highly correlated with BS reflectance readings. Other non-black, non-carbon particles also have optical properties such that they can affect the reflectance readings (Pedace and Sansone, 1972).

Since highly variable relative proportions of atmospheric carbon and non-carbon PM can exist from site to site or from one time to another at the same site, then the same absolute

BS reflectance reading can be associated with markedly different amounts (or mass) of particles or, even, carbon collected. Site-specific calibrations of reflectance readings against actual mass measurements from collocated gravimetric monitoring devices are therefore necessary in order to obtain approximate estimates of atmospheric PM concentrations based on the BS method. A calibration curve relating mass or atmospheric PM concentration (in $\mu g/m^3$) to BS reflectance readings obtained at a given site may serve as a basis for crude estimates of PM (mainly small particle) levels at that site over time, so long as the chemical composition and relative proportions of elemental carbon and non-carbon PM do not markedly change.

For British National Survey and OECD work site-specific BS mass calibration curves were determined in the 1960s for numerous urban areas in the United Kingdom and Europe, and such curves were interrelated or normalized to define certain "standard" curves. Two standard calibration curves were adopted: (1) a British standard smoke curve defining relationships between PM mass and BS reflectance readings for London's atmosphere in 1963, used to yield BS concentration estimates (in $\mu g/m^3$) reported in many published British epidemiological studies; and (2) an OECD international standard smoke curve, against which smoke reflectance measurements made elsewhere in Europe were compared to yield smoke concentration estimates (in $\mu g/m^3$) reported in European studies on PM effects. Of crucial importance in assessing such studies is the fact that the actual PM mass or smoke concentration at a particular site may differ markedly (e.g., by factors of two or more) from the corresponding mass or concentration (in $\mu g/m^3$) associated with a given reflectance reading on either of the two standard curves;[a] and great care must be applied in interpreting what any reported BS value in $\mu g/m^3$ means at all. Further complicating interpretation of smoke data used in most epidemiological studies is the lack of reporting of specific quality assurance information for cited aerometric measurements. Such information has only been reported (Warren Spring Laboratory 1961, 1962, 1966, 1967, 1972, 1975; OECD 1964; Moulds, 1962; Ellison, 1968) in general terms for United Kingdom National Survey data used in some British studies.

The ASTM or AISI light transmittance method is similar in approach to the British smoke technique. The instrument has a D_{50} cutpoint of ≅5 μm and utilizes an air flow intake apparatus similar to that used for the BS method, depositing collected material on a filter paper tape periodically advanced to allow accumulation of another stain over a standard time period. Opacity of the stain is determined by transmittance of light through the deposited material and filter paper, with results expressed in terms of optical density or coefficient of haze units (CoHs) per 1000 linear feet of air sampled rather than in terms of mass units. Thus, CoHs readings roughly index the soiling capacity of PM in the air and, like BS readings, are most strongly affected by fine-mode elemental carbon particles. Coefficient of haze readings, however, are more markedly affected by non-carbon particles than BS measurements. The ASTM method does not directly measure mass or determine chemical composition of the PM collected. Attempts to ever relate CoHs to $\mu g/m^3$ would require site-specific calibration of CoHs readings against mass measurements determined by a collocated gravimetric device, but the accuracy of such mass estimates would be subject to question. Only one attempt at calibration of CoHs readings against corresponding particle mass levels has been reported (Ingram, 1969; Ingram and Golden, 1973), for only one city (New York). The calibration results, however, are applicable in only a limited fashion to New York City aerometric data.

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[a] For this reason, smoke data reported in $\mu g/m^3$ based on either the British or OECD Standard curve are generally most appropriately interpreted in terms of "nominal" $\mu g/m^3$ smoke units and cannot be accepted as accurate estimates of airborne PM mass unless corroborated by local site-specific gravimetric calibrations. In other words, unless based on local site-specific calibrations, smoke readings in $\mu g/m^3$ cannot yield quantitative estimates of atmospheric PM concentrations. Otherwise, such readings only allow for rough qualitative (i.e. <; =; or >) comparisons of amounts of PM present at a given time versus another time at the same site and do not permit meaningful comparisons between PM levels at different geographic areas having airborne PM of different chemical composition (especially in terms of relative proportions of elemental carbon).

The high-volume (hi-vol) sampler method more widely used in the United States to measure total suspended particulate matter (TSP) collects particles on a glass-fiber filter by drawing air through the filter at a flow rate of approximately 1.5 m^3/minute, thus sampling a higher volume of air per unit of time than the above PM sampling methods. This permits collection of sufficient PM in a 24-hour period to allow for direct weighing and chemical analysis of sampled material. The high flow rate, the geometric shape of the shelter housing the sampler, and other features of standard types of apparatus used result in sampling efficiency varying with windspeed. The D_{50} cutpoint for the hi-vol sampler is typically around 25 to 50 μm and collection of larger particles tends to drop off rapidly above such cutpoints (Wedding et. al., 1977; McFarland and Ortiz, 1980). Thus, the hi-vol sampler, as typically employed, collects both fine- and coarse-mode particles that may include windblown crustal material of natural origin (especially in dry rural areas). Only rarely have cyclone samplers or other variations of the hi-vol sampler with smaller size cutpoints been utilized in epidemiological studies to limit collected particles to an inhalable range, but even then the cutpoints achieved were not sharp or independent of windspeed. Numerous factors other than windspeed, as discussed in Chapter 3, can affect PM measurements by hi-vol sampling techniques. However, quality assurance information for TSP measurements reported in most American epidemiological studies is largely lacking, except for information on U.S. Environmental Protection Agency CHESS Program data (U.S. Congress, House of Representatives, 1976).

Among the consequences of the broader size range of particles sampled by the hi-vol method versus the BS or ASTM methods are severe limitations on intercomparisons or conversions of PM measurements by those methods to equivalent TSP units or vice versa. No consistent relationship was found, for example, between BS and TSP measurements taken at various sites or during various seasons at the same site (Commins and Waller, 1967; Lee et al., 1972; Ball and Hume, 1977; Holland et al., 1979). Some exceptions exist, e.g., during severe London air pollution episodes, when low wind speed conditions resulted in settling out of larger coarse-mode particles and marked increases in fine-mode particles to constitute most of the PM present.

Then TSP and BS levels (in excess of ≅ 500 μg/m^3) tended to converge, as would be expected when both methods are essentially sampling only fine-mode particles (Holland et al., 1979).

Taking into account the foregoing information on SO_2 and PM measurement methods and factors affecting the quality of results obtained with routine field monitoring, aerometric data cited in various epidemiological studies must generally be viewed as providing at best only very approximate estimates of atmospheric levels of sulfur dioxide, other sulfur compounds, or other PM associated with reported health effects. Further, to the extent that the aerometric data cited are derived from use of techniques with limited specificity for the substance(s) purportedly measured or the relative contributions of sulfur oxides or PM to observed health effects cannot be distinguished from each other or from the effects of other covarying pollutants, then the aerometric data and associated health effects reported might be more appropriately viewed as relatively nonspecific indicators of the effects of overall air pollutant mixtures containing sulfur oxides and PM.

14.3 ACUTE PARTICULATE MATTER/SULFUR OXIDES EXPOSURE EFFECTS

14.3.1 Mortality

14.3.1.1 Acute Episode Studies--Detailed study of human health effects associated with episodes of severe air pollution spans a period of less than 50 years. The earliest reliable documentation of such episodes describes a 1930 incident in the Meuse Valley of Belgium. Dense fog covered the valley from Liege to Huy (Firket, 1931, 1936) from December 1 to 5, 1930, accompanied by an anticyclonic high pressure area with low winds and large amounts of PM. Approximately 6,000 residents in the valley became ill and 60 deaths associated with the fog occurred on December 4-5. The people who died were sick for only a short time and the on-set of acute illnesses abated rapidly when the fog dispersed. Although no other immediate deaths occurred, several persons affected by the fog died much later from complications associated with fog-induced injuries. The death rate in the area was 10.5 times normal.

A similar event later occurred in Donora, Pennsylvania (Shrenk et al., 1949). Donora was blanketed by a dense fog during October 1948, which adversely affected 43 percent of the population of approximately 10,000 people. Twenty persons, mostly adults with preexisting cardiopulmonary diseases, died during or shortly after the fog due to cardiorespiratory causes; and 10 percent of the population was classified as being severely affected. No pollution measurements were made during the incident, but during subsequent inversion periods in the same area, presumably not as severe in pollutant elevations as the one in October 1948, daily averages of SO_2 as high as 0.4 ppm (~1140 μg/m^3) were recorded. In a follow up study of Donora, Ciocco and Thompson (1961) found increased mortality rates and morbidity effects (e.g., heart disease and chronic bronchitis) among those residents who reported acute illness during the 1948 episode in comparison to those reporting no acute illness. The Meuse Valley and Donora incidents demonstrated that severe air pollution can cause death and serious morbidity effects in exposed human populations and raised the possibility of PM and SO_2 being among air pollutants contributing to the induction of such health effects.

As shown in Table 14-1, a series of episodes was also documented in London between 1948 and 1962 (Ministry of Health, 1954; Scott, 1953; Logan, 1953; Wilkins, 1954a,b; Hewitt, 1956; Gore and Shaddick, 1958; Burgess and Shaddick, 1959; Martin and Bradley, 1960; Clifton et al., 1960; Lawther, 1963). Excess mortality reported during those episodes occurred mainly among the elderly and chronically ill adults during periods of prolonged marked elevations in air pollution lasting for several days. Various factors have been discussed which might help to explain some of the excess mortality (Holland et al., 1979), including possible influences not only of increased air pollution but also of high humidity (fog) and low temperatures. Regardless of the relative contributions of these different factors, there exists a clear consensus that increases in mortality were associated with air pollution episodes when 24-hour concentrations of both SO_2 and BS exceeded 1000 $\mu g/m^3$ in London (Rall, 1974; Higgins, 1974; Goldsmith and Friberg, 1977; NRC/NAS, 1978a,b; Shy et al., 1978; Holland et al., 1979; WHO, 1979; Shy, 1979). The available data, however, do not allow for clear delineation of the effects of specific pollutants acting alone or in combination.

TABLE 14-1. EXCESS DEATHS AND POLLUTANT CONCENTRATIONS DURING SEVERE AIR POLLUTION EPISODES IN LONDON (1948 to 1962)

Date	Duration (days)	Deviation from $\bar{X}$ of total excess deaths[a]	Maximum 24-hr pollutant concentration, $\mu g/m^3$ Smoke (BS)	Maximum 24-hr pollutant concentration, $\mu g/m^3$ SO_2 (H_2O_2 titration)
Nov. 1948	6	750	2780	2150
Dec. 1952	4	4000	4460[b]	3830
Jan. 1956	4	1000	2830	1430
Dec. 1957	4	750	2417	3335
Jan. 1959	6	250	1723	1850
Dec. 1962	5	700	3144	3834

[a] Note that the numbers of excess deaths listed represent 15 to 350 percent increases in normal London baseline death rates during the years listed.

[b] Note that peak and 24-hr BS levels were likely much higher than 4460 $\mu g/m^3$ due to rapid saturation of filter paper by collected PM.

Acute episodes of high air pollution have also occurred in the United States since the 1948 Donora episode, but no single event reached the magnitude of the London episodes. Some published studies (Greenburg et al. 1962, 1967; McCarroll and Bradley, 1966; Glasser et al. 1967), for example, suggested that increases in mortality and morbidity may have occurred during some New York City episodes in the 1950s and 1960s, when PM levels exceeded 5.0 to 8.0 CoHs and SO_2 exceeded 1000 $\mu g/m^3$ (0.40 ppm), as measured at a single monitoring station in central Manhattan. The earliest episode (November, 1953) involved a prolonged (10 to 12-day) anticyclonic temperature inversion, during which PM levels built up to peak 24-hour levels of 6.0 to 8.0 CoHs on the last 4 days and SO_2 levels (estimated by "total acidity") reached hourly peaks of 0.47 to 0.80 ppm (1231 to 1800 $\mu g/m^3$) on each of those days. Under these conditions of simultaneous elevations of PM and SO_2, average daily death rates were estimated to have increased by 18 (8 percent) over a control total mortality comparision level of 226/day distributed equally across all age groups. However, close inspection of the published data does not convincingly reveal the reported total mortality increases; nor do the published cause-specific mortality data suggest notable increases in specific death categories plausibly affected by air pollution.

Later, in November-December, 1962, during an episode of intermittent daily peaks of PM exceeding 5.0-6.0 CoHs and SO_2 peaking at 0.4-0.5 ppm (1050 to 1310 $\mu g/m^3$) at the same central New York City sampling station, no increased death rates were detected by Greenburg et al (1963); nor were clinic visits at four New York City hospitals for cardiopulmonary conditions increased. Only the daily number of upper respiratory complaints appeared to increase significantly ($P < .01$) among elderly residents in four old-age homes. In contrast, significant increases in respiratory morbidity and mortality among older (45 to 65 and > 65 years) age groups were reported (Greenburg, et al. 1967) to have occurred during January 29 to February 12, 1963, when PM and SO_2 intermittently peaked at daily levels in excess of 5.0 to 6.0 CoHs and 0.4 to 0.8 ppm (1050 to 1800 $\mu g/m^3$), respectively. During this period of high PM and SO_2 pollution, coincident with the occurrence of the coldest New York City temperatures in decades and an influenza epidemic, Greenburg et al. (1967) estimated that 200 to 400 excess deaths (4 to 20 percent increases) occurred in comparison to various control baseline daily mortality values. However, increased death rates did not appear on or immediately after all of the days of peak PM and SO_2 levels during the January-February, 1963 episode or on other scattered days of comparably high or higher PM or SO_2 peaks in the weeks immediately preceding or following the episode period. The data reported, therefore, provide only a very weak basis upon which to assert that excess mortality attributable to air pollution was superimposed upon New York City death rates already elevated by cold weather and influenza in early 1963.

Somewhat more convincing are elevated mortality levels observed (especially in those 65 years or older) during the 1966 Thanksgiving Day weekend as reportedly occurred in the absence of cold weather (temperatures = 34° min to 64°F max) or epidemic illnesses (Glasser et al. 1967). Daily mean PM levels remained in the range of 5.0 to 6.0 CoHs for 3 successive days (November 23 to 25) during which time daily SO_2 levels averaged 0.4 to 0.5 ppm (1050 to 1310 $\mu g/m^3$) and mean daily total mortality levels reportedly exceeded those (237/day) of comparison control periods by 24 (10 percent). Other comparable increases in daily mortality appeared to occur in association with upward excursions of PM to peak hourly levels > 5.0 CoHs during 2-3 successive days 2 weeks prior to the November 23 to 25 episode and on 2 days about 2 weeks after the episode. (Note that daily mean and hourly peak SO_2 levels did not exceed 0.3 ppm (790 $\mu g/m^3$) during the high PM days in early November.) Also, cause-specific mortality for certain cardiovascular diseases appeared to rise noticeably over control levels on or immediately after these same days of elevated PM levels, as well as during the November 23 to 25 episode. These overall results tend to suggest that the elderly and individuals with prexisting cardiovascular disease may have been adversely affected by closely occurring instances of prolonged periods of severe air pollution containing high concentrations of PM and/or SO_2 in late 1966.

Independent evaluations of the same New York City data on mortality and air pollution relationships initially led to a published report (McCarroll and Bradley, 1966) confirming apparent associations between increased mortality and acute episodes of high PM and SO_2 levels. However, later reexamination of the New York City data and the published analyses by the Greenburg group and by McCarroll and Bradley (1966) led Cassell et al. (1968) to question the validity of the earlier published conclusions, especially in view of difficulties in separating air pollution episode effects on mortality from effects of competing factors such as temperature and humidity extremes and epidemic illnesses, which appeared to exert much larger effects on death rates than the air pollution episodes. Still further doubts regarding the reported New York City air pollution episode-mortality associations are raised by inconsistencies in the data, such as no evident mortality increases being associated with some days of PM and/or SO_2 elevations as high or higher than those on other days reported to be associated with excess mortality. Only in November-December, 1966, did there seem to exist some possible hints of consistent associations between high pollutant elevations and excess mortality over control comparison levels. Thus, the results of the New York City episode studies, while not necessarily inconsistent with the hypothesis of increased mortality being associated with episodic elevations of PM and SO_2, do not provide strong evidence in support of the hypothesis either. At best, the studies might suggest possible increases in mortality and respiratory morbidity (mainly among older members of the population and those with preexisting cardiovascular or respiratory diseases) under unusual conditions of prolonged multiday periods of severe air pollution containing simultaneously high concentrations of PM and SO_2 likely in excess of 5.0 to 8.0 CoHs and 1000 $\mu g/m^3$, respectively (see comments in Section 14.3.1.2 and Appendix 14A on limitations concerning quantitative statements that can be made based on aerometric data from the single central Manhattan monitoring station used to index PM and SO_2 concentrations for the entire New York City area).

When a marked increase in air pollution is associated with a sudden rise in the death rate or illness rate that lasts for a few days and both return to normal shortly thereafter (as documented in some of the above studies), a causal relationship is strongly suggested. Sudden changes in weather, however, which may have caused the air pollution incidents, must also be considered as another possible cause of the death rate increase. On the other hand, the consistency of associations between SO_2 and particulate matter elevations in London and increases in mortality make it extremely unlikely that weather changes alone provide an adequate explanation for all such observations. This view is further reinforced by: (1) the fact that at least some episodes were not accompanied by sharp falls in temperature; and (2) other weather changes of similar magnitudes to those accompanying the above pollution episodes are not usually associated with such dramatic increases in mortality in the absence of greatly increased levels of SO_2, particulate matter, or other pollutants. In summary, the above London episode studies appear to provide clear evidence for substantial increases in excess mortality when the general population was exposed over several successive days to air pollution containing SO_2 concentrations $\geq$ 1000 $\mu g/m^3$ in the presence of PM levels over 1000 $\mu g/m^3$ (BS). Certain New York studies also tentatively suggest that small increases in excess mortality may have occurred at simultaneous elevations of 1000 $\mu g/m^3$ SO_2 and PM above 5.0 - 8.0 CoHs, but this is much less clearly established.

Comparision of the New York City episode data and those for the Meuse Valley, Donora, and London episodes may reveal further important information. Perhaps most striking are the much lower estimates of excess mortality reported for the New York episodes (at most 4 to 20%) compared to the 15- to 350-percent death rate increases during the London episodes and even larger mortality rate increases in Donora and the Meuse Valley. Numerous factors might be cited to explain the striking differences, including likely variations in the specific chemical composition of the mixes of pollutants present in the different areas and the much greater peak levels of pollutants (including PM and/or SO_2) that were probably present during the non-New York episodes. Also of probable considerable significance are two other features typifying the episodes in the Meuse Valley, Donora, and London: (1) the presence of extremely dense fog together with accumulating air pollutants, possibly providing the basis for transformation of pollutants to potentially more toxic forms (e.g. formation of sulfuric acid aerosol or absorption of PM into water droplet particles) resulting in more deposition of toxic substances in tracheobronchial regions of the respiratory tract (see Chapters 11 and 13); and (2) the generally much more prolonged, continuous exposures of the non-New York populations to marked elevations of the pollutants. Examination of the published New York City episode reports reveals that during such episodes the contributing temperature inversion conditions typically intensified during evening hours (accumulating air pollutants over night) but dissipated during the morning hours, resulting in invariably much higher peaks in PM and SO_2 in the morning than in the afternoons (when PM and SO_2 levels fell back to near normal levels). This is in contrast to the continuously high pollutant levels that apparently persisted for several ($\geq$ 4) successive days during the Meuse Valley, Donora, and London episodes, with the largest increases in mortality tending to occur on the later days of each episode. Duration of exposure, even at the extremely high levels of pollutants present, and the presence of certain other interacting factors such as high humidity (fog), then, appear to be important determinants affecting increases in mortality observed in acute episode incidents. Such factors must, therefore, be taken into account as important limiting considerations in attempting to generalize or extrapolate observed episodic air pollution-mortality dose-response relationships from one geographic location (or time period) to another.

14.3.1.2 Mortality Associated with Non-Episodic Variations in Pollution--A number of reports have investigated relationships between mortality and air pollution in England during periods with no unusual air pollution episodes (Gore and Shaddick, 1958; Burgess, 1959; Clifton et al., 1960; Martin and Bradley, 1960; Scott, 1963; Lawther, 1963; Martin, 1964; Waller et al., 1974). For most of these studies, 15-day moving averages were constructed and the effects of pollution were assessed in terms of daily deviations from these baselines. Lawther (1963) reported that increases in daily deaths during the winter of 1958-59 appeared to be associated with 24-hr concentrations of BS >750 μg/m^3 and SO_2 >715 μg/m^3 (0.25 ppm) during a long (59-day) period of thick fog. Further, Lawther (1963) reported that increases in daily deaths generally did not seem to be associated with pollutants at lower concentrations during 1958-59; nor did they occur at similar pollutant levels during the prior winter having only 8 days of fog, suggesting again the possible importance of high humidity conditions as a key factor interacting with air pollutants to increase mortality. Similar studies in Sheffield (Clifton et al., 1960) did not yield confirmatory results; that is, while increases in deaths appeared to be possibly associated with very high concentrations of pollutants, random variations in the number of deaths were so large that firm conclusions could not be drawn.

Probably the most important British studies of mortality associated with nonepisodic exposure to sulfur oxides and PM are those of Martin and Bradley (1960) and Martin (1964). The first of these studies related daily mortality from all causes and from bronchitis and pneumonia to the level of SO_2 and smoke in London during the winter of 1958-1959, as measured by a multistation air monitoring network operated throughout London. The authors found a considerable number of coincident peaks in pollution level and daily mortality. The correlation of mortality from all causes with pollutants measured on the log scale was 0.613 for smoke (BS) and 0.520 for SO_2. Martin and Bradley (1960) reported that neither temperature nor humidity was significantly correlated with London mortality studied during the winter of 1958-59, noting a very low correlation coefficient (r = -0.030) for temperature and deaths for the entire 1958-59 winter and the occurrence of several peaks in mortality during November and December, 1958, when temperatures were substantially over 38°F. They further noted that a range of 30-38°F is characteristic of most winter fogs and temperatures consistently below 30°F (when temperature effects on mortality can be expected) are the exception. Though the authors emphasized the relationship between change in pollution level and number of deaths and lack of meteorological effects, an influenza epidemic during part of the study may have influenced some of the results.

Martin and Bradley (1960), however, reported number of deaths, smoke levels, and SO_2 levels from November 1, 1958 to February 28, 1959 allowing independent analysis of the data. One such independent analysis was performed by Ware et al. (1981) but excludes the month of February, in which the epidemic of Type A influenza also significantly influenced daily mortality. For the remaining 92 days, the deviations of daily mortality from the 15-day moving average (truncated at each end of the series) were computed and appeared to show a consistent and significant trend of increasing mortality with increasing BS and SO_2 levels. The results of the Ware et al. (1981) analysis are graphically depicted in Figure 14-1. Mean deviations from the 15-day moving average changed from negative to positive between 500-600 μg/m^3 BS and 300-400 μg/m^3 SO_2, which are concentrations at which apparent marked increases in mortality occurred in comparison to lesser increments seen at lower pollutant levels. However, the 500-600 μg/m^3 BS and 300-400 μg/m^3 SO_2 levels cannot be accepted as clearcut thresholds for

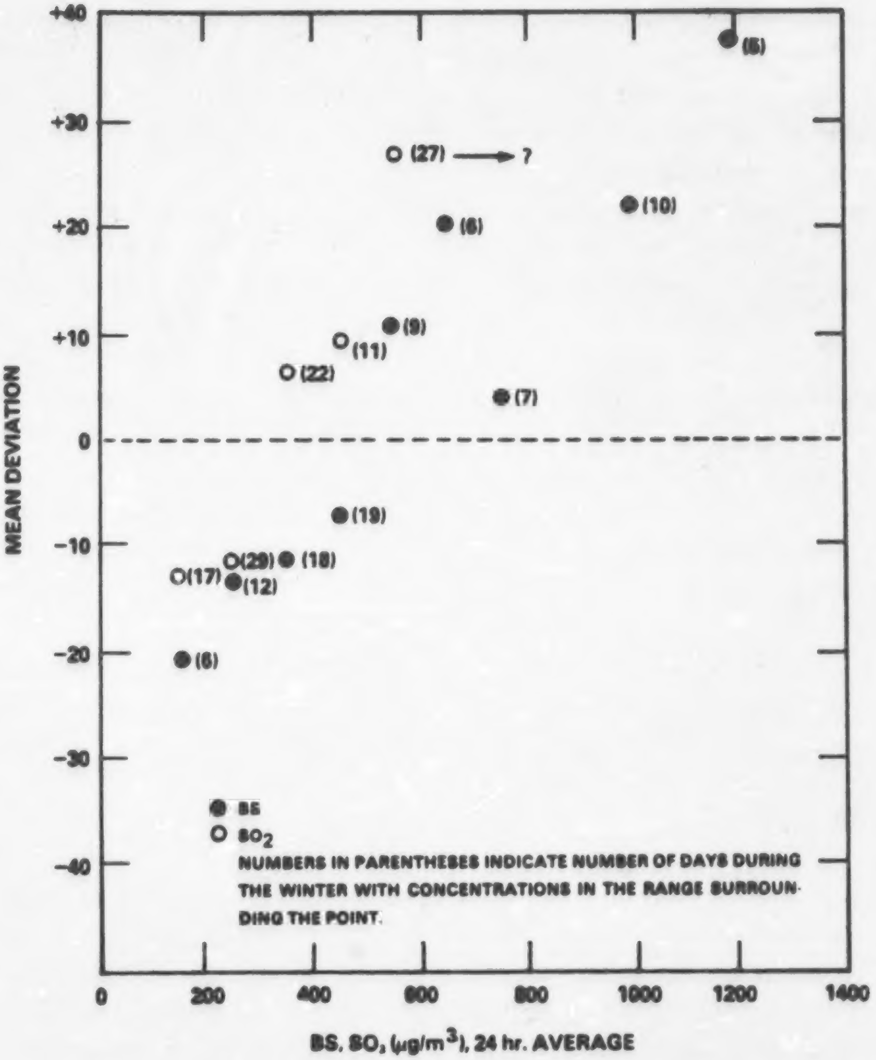

Figure 14-1. Martin and Bradley (1960) data as summarized by Ware et al. (1981) showing average deviations of daily mortality from 15-day moving average by concentration of smoke (BS) and SO_2 (London, November 1, 1958 to January 1, 1959). As daily pollutant levels decrease, the positive mortality deviations also decline. At still lower pollution levels, daily mortality deviations fall increasingly below the 15-day moving mean, suggesting the possibility of a continuum of exposure-response over the range examined. Note that the highest value depicted for mortality associated with SO_2 levels perhaps should more appropriately be shifted to some point to the right as indicated above by→?.

mortality, given that the dose-response curves plotted in Figure 14-1 are highly suggestive of a possible monotonic relationship across the entire range of exposure values shown. In fact, it is unclear at what level significant excess mortality first occurred, although graphical presentation (Figure 14-1) of the Martin and Bradley (1960) data as tabularly aggregated by Ware et al. (1981) does indicate that notable increases clearly occurred at BS and SO_2 levels somewhere in the range of 500-1000 $\mu g/m^3$ and small increases in mortality may have occurred at levels below 500 $\mu g/m^3$ of either pollutant.

Although the graphical plot of the Martin and Bradley (1960) data in Figure 14-1 suggests a monotonic relationship across the entire range of exposure values, adequate evaluation of this possibility would require more thorough statistical analyses, including consideration of possible contributions of temperature and humidity, autoregression tendencies, and other sources of error to the observed mortality patterns. Thus, whereas temperature and humidity were reported by Martin and Bradley (1960) not to be significantly correlated with daily mortality, both pollution levels and daily mortality increased throughout the period of study, such that the possibility of other extraneous seasonal variables contributing to the mortality increases cannot be ruled out. Also the above analyses of dose-response relationships by Martin and Bradley (1960) and Ware et al. (1981) were both conducted assuming no error in the PM and SO_2 concentrations used to estimate population exposure levels. The air pollutant estimates used by them represent average BS and SO_2 levels as monitored at seven air sampling stations located at widely dispersed sites in various areas of London. Several types of possible error or variation in BS and SO_2 data could be associated with the 7-site average concentration estimates reported in $\mu g/m^3$ by Martin and Bradley (1960). Of most crucial importance are: (1) the accuracy and precision of site-specific calibrations by which estimates of PM mass (in $\mu g/m^3$) were derived from BS reflectance readings; (2) errors associated with field applications of BS and SO_2 measurement methods at the monitoring sites; and (3) the variation or range of specific PM or SO_2 values obtained at the different sites from which daily mean concentration exposure estimates for all of London were derived.

Little information exists by which to judge the error associated with the first factor listed above, other than to note that site-specific calibrations of PM mass ($\mu g/m^3$) against BS reflectance readings carried out in 1956 at a central London site, as described by Waller (1964), appear to confirm reasonably well the BS mass (in $\mu g/m^3$) to reflectance calibration (D.S.I.R.) curve employed in estimating mass from reflectance readings at the above seven London sites in 1958-59 and for several more years until 1963. By 1963, however, the mass to reflectance relationship defined by calibrations at another nearby London location (Waller, 1964) appeared to have shifted somewhat from the 1956 curve (see Figure 14C-2; Appendix 14C). This possibly reflected changes in chemical composition of the ambient airborne PM as the proportion of elemental carbon present declined due to reductions in emissions of sooty incomplete coal combustion products as "Clean Air Zones" were established following enactment of the British 1956 Clean Air Act.

In regard to the second factor listed above, little specific information is presently accessible by which to ascertain with confidence the accuracy or precision of PM or SO_2 measurements taken at each of the seven monitoring sites more than 20 years ago. Only limited quality assurance information on sources of error related to measurements of the same type taken as part of the British National Air Pollution Survey exists by which to speculate on possible measurement errors associated with the London aerometric data reported by Martin and Bradley (1960). Examples of such quality assurance information are summarized in Appendix 14C; but Warren Spring Laboratory has commented that such measurement errors generally tend to cancel each other out, so that the precision of the PM and SO_2 measurements are generally assumed to be within about 6 percent (based on several intercomparison tests; see British Standards Institution, 1964) in the absence of evidence to the contrary suggesting systematic measurement error or bias in specific sampling data.

In regard to assessing the third source of possible error in the London exposure estimates, the original BS and SO_2 data for the seven monitoring stations used to derive daily London-wide average BS and SO_2 levels reported by Martin and Bradley (1960) were obtained by EPA. Calculation of pair-wise correlation coefficients revealed very high correlations (all $\geq$ 0.50; most $\geq$ 0.75) between: (1) daily BS values reported at any two individual sites, even the two most widely separated geographically; (2) daily SO_2 levels at any two specific sites; and (3) BS and SO_2 readings at the same site or between any two given sites. The absolute amount of variation of any individual site BS or SO_2 readings from the daily 7-station mean increased as a function of increasing concentrations but, in general, the 7-station average seems to be an excellent index of city-wide increases or decreases in airborne PM or SO_2 levels. The daily London mortality data points reported by Martin and Bradley (1960) for the 1958-59 winter (excluding the influence of the epidemic period data in January-February, 1959) were then plotted in relationship to the seven-station averages of PM and SO_2 as recalculated from the original seven-station data. These relationships are depicted in Appendix D (Figure 14D-3 for BS values and Figure 14D-6 for SO_2 values), along with non-linear fitted curves and two sets of 95 percent confidence intervals, one set assuming error only in the mortality estimates and a wider set of upper and lower 95 percent confidence bounds reflecting additional error or variation in the PM and SO_2 exposure estimates. Information on the derivation of the fitted curves and confidence intervals is also presented in Appendix 14 D. Examination of results depicted in Figures 14D-3 and 14D-6 reveals very small predicted increments in total mortality over estimated 1958-59 London winter background levels of about 280 deaths/day (at < 150 $\mu g/m^3$) as a function of increasing BS or SO_2 levels up to 500 $\mu g/m^3$. However, increasingly more marked increases in daily total mortality over background levels become apparent as BS concentrations rise from 500 to 1000 $\mu g/m^3$ and clearly large incremental increases are apparent at successively higher BS levels over 1000 $\mu g/m^3$. Given the relatively wide confidence bounds associated with these particular data, however, much caution should be employed in interpreting reported dose-response relationships as depicted in Appendix 14D or as derived by other analyses of these data; and such analyses should not be taken as demonstrating precise quantitative relationships for uncaveated use for risk assessment purposes.[a]

An analysis similar to that used by Martin and Bradley (1960) was carried out by Martin (1964) for the winter of 1959-60. This winter had fewer incidents of high pollution. The significant positive correlation between mortality and pollution was, however, still present although the coefficients were somewhat lower than in the previous year. The Martin (1964) results were based on analyses combining high pollution days from 1958-59 and 1959-60, after excluding days on which pollution had fallen from a previously higher level. The mean deviation was positive for every BS level above 500-600 $\mu g/m^3$ and SO_2 level above 400-499 $\mu g/m^3$, but again no clear threshold for significant increases in mortality could be clearly delineated. However, the most marked increases occurred for BS levels over 1200 $\mu g/m^3$ and SO_2 levels exceeding 900 $\mu g/m^3$. Considerable covariation in levels of the two pollutants, however, preclude attribution of the apparent mortality effects to either one alone. Bronchitis mortality was also significantly, though less strongly, correlated with pollution level, but pneumonia mortality was not correlated with pollution.

The above discussions of the Martin and Bradley (1960), Martin (1964), and Ware et al. (1981) analyses, it should be noted, concern a relatively small number of observations on air pollution-mortality relationships obtained during just two London winters (1958-59; 1959-60). Thus, while the results of these analyses are highly suggestive of certain relationships between London daily mortality rates and BS or SO_2 levels, analyses of more London mortality data from additional years would be valuable in order to assess the strength and representativeness of associations detected for the 1958-59 and 1959-60 London winters.

Such analyses of London nonepisodic air pollution-mortality relationships over a much longer time-period have recently been reported by Mazumdar et al. (1981) in the published proceedings of an American Air Pollution Control Association conference. Mazumdar et al. (1981) employed two different types of statistical approaches in analyzing excess mortality over 15-day moving averages for London winters from 1958-59 to 1971-72. An analysis was initially used whereby variations in nonepisodic excess mortality were analyzed in relation to pollutant levels categorized according to quartiles (i.e. 4 quartiles of BS values versus 4 SO_2 quartiles). This allowed for statistical comparison of mortality for cells corresponding to highest BS quartile values versus lowest SO_2 quartile levels and so on, as well as other combinations of BS and SO_2 quartile values including 4 quartiles of SO_2 levels "nested" within each quartile of BS levels. Mazumdar et al. (1981) reported that, by these analyses, they were able to separate out the relative contributions to mortality of BS and SO_2 (which closely covaried in London during most years studied) and that significant relationships existed between excess mortality and BS levels but not between mortality and SO_2 levels during the 1958-59 to 1971-72 London winters. Serious questions can be raised regarding specific details concerning the quartile analyses used and the validity of reported conclusions regarding the separation of BS from SO_2 effects. For example, the numbers of data points falling in the highest BS-lowest SO_2 and lowest BS-highest SO_2 quartile cells are extremely small (being only 2 and 1 respectively), and not likely allowing for any reasonable statistical comparisons against other quartile combinations. However, based on their conclusions derived from the quartile analyses, Mazumdar et al. (1981) attempted to further define possible dose-response relationships between excess mortality and BS concentrations by means of regression analyses.

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[a] It should also be noted that the analyses presented in Appendix 14D were not adjusted for temperature, humidity, temporal trends or possible autoregression factors. Another recent, unpublished analysis of the data taking such factors into account is summarized in Appendix 14E for informational purposes and appears to show small incremental increases in mortality being associated with BS levels in the range of 150-500 $\mu g/m^3$ during the 1958-59 London winter.

Mazumdar et al. (1981) used both a linear and a non-linear (quadratic) model in carrying out their regression analyses on the 1958-59 to 1971-72 London winter mortality data. Figure 14-2 depicts the dose-response relationships defined by each of the two models, plotted on the scatter diagram for excess mortality data versus BS levels for all of the winters analyzed. A clearer depiction of the hypothesized dose-response relationships obtained with the linear and quadratic models is presented in Figure 14-3. Both analyses (linear and quadratic) indicate that small increases in mortality were associated with London PM levels in the range of 150-500 $\mu g/m^3$ BS and more marked mortality increases occurred as BS levels rose to 500-1000 $\mu g/m^3$ or more. The findings of mortality being significantly associated with the lower range of BS values (150-500 $\mu g/m^3$) were further confirmed by analyses of mortality rates occurring only on days when BS levels did not exceed 500 $\mu g/m^3$. Importantly, temporal factors as well as temperature and humidity effects were taken into account as part of the regression analyses employed. However, shifts in the specific calibration curves used to relate BS reflectance readings to estimates of PM mass (in $\mu g/m^3$) were not taken into account in the analysis. Prior to 1963, the D.S.I.R. curve alluded to earlier (page 14-18) was used in determining BS mass estimates from reflectance readings in London; after 1963-64, the British National Air Pollution Survey standard curve was used instead, representing calibrations between PM mass ($\mu g/m^3$) and BS reflectance readings in London in 1963. Because of differences between the D.S.I.R. curve and the 1963 curve, as well as the likely decreasing applicability of the 1963 curve for each successive subsequent year, probably a more appropriate statistical approach would be to analyze the London mortality-air pollution data on a year-by-year basis. This would allow for better determination of the consistency of any significant dose-response relationships for any given year(s) versus findings for other years.

Inspection of London winter mortality and pollution data tabulated by Mazumdar et al. (1981) for separate years indicates both higher mortality rates as well as PM and SO_2 levels for 1958-59 than for all subsequent winters other than 1962-63. It is not clear as to what interacting factors may distinguish 1958-59 (and perhaps 1962-63) as winters having apparently higher mortality rates associated with increased PM and/or SO_2 levels. Possibly, consistent with hypotheses noted earlier, more frequent occurrences of high humidity (fog) days in conjunction with elevated pollutant levels during those two winters may offer one plausible

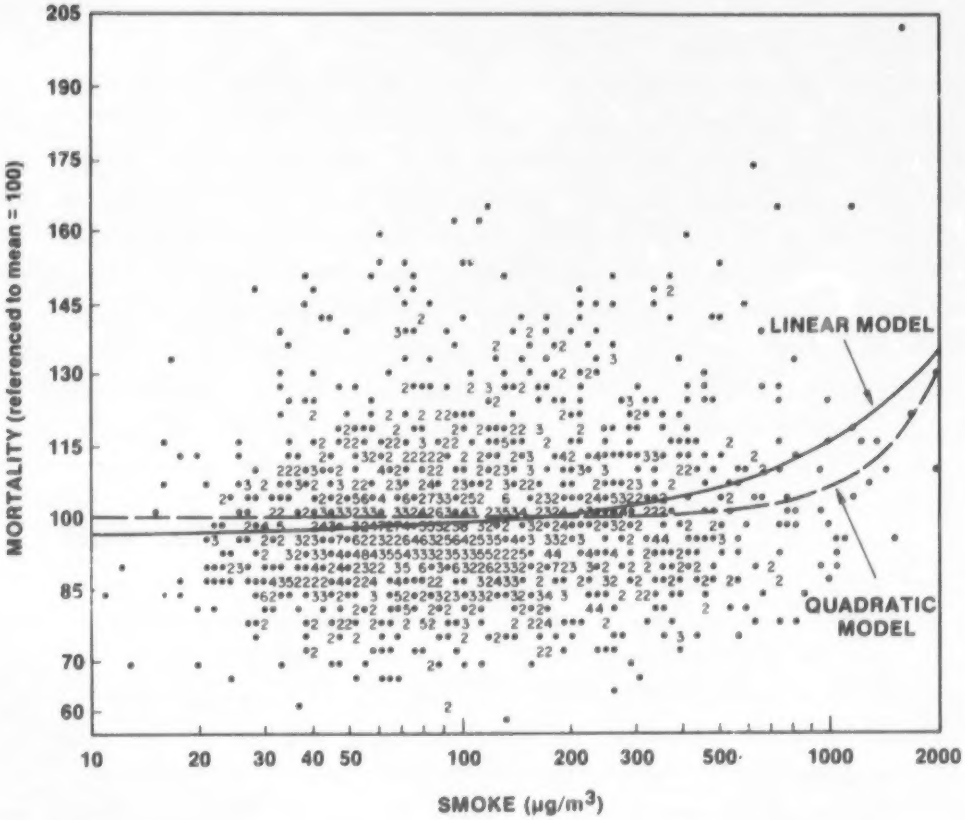

Figure 14-2. Linear and quadratic dose-response curves plotted on the scattergram of mortality and smoke for London winters 1958/59 to 1971/72. Source: Mazumdar et al. (1981).

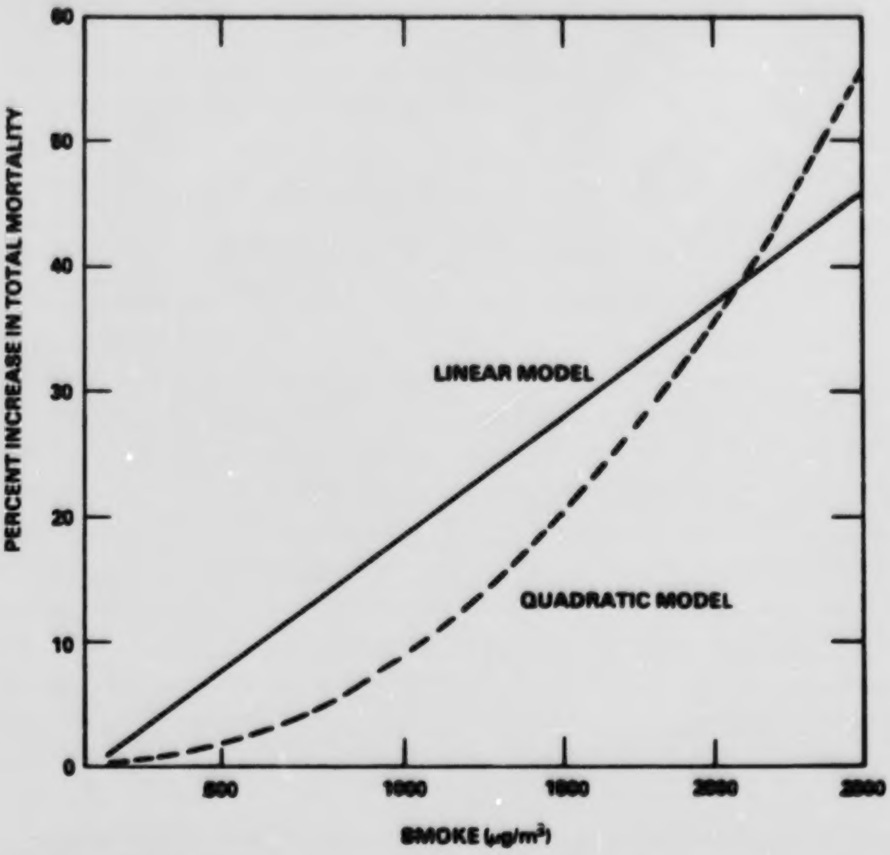

Figure 14-3. Hypothetical dose-response curves derived from regressing mortality on smoke in London, England during winters 1958/59 to 1971/72. Results obtained with linear (—) and quadratic (- -) models are depicted for comparison.
Source: Mazumdar et al. (1981).

explanation; but more detailed analyses of the London data on a year-by-year basis would be required in order to assess this possibility.[a]

It should be noted that, although the above analyses point to significant associations between mortality and PM or SO_2 levels in London, the relative separate contributions of PM (as BS) or SO_2 cannot be clearly determined based on the available data; nor can the possibility be ruled out that the observed mortality associations are due to other, unmeasured covarying specific air pollutants or the total pollutant mix for which BS or SO_2 may be serving as excellent indices or surrogates.

A number of investigators have also evaluated possible relationships in the United States (especially in New York City) between mortality and daily variations in PM or SO_2 air pollution during non-episodic periods (Hodgson, 1970; Glasser and Greenburg, 1971; Schimmel and Greenburg, 1972; Lebowitz, 1973a,b; Buechley et al., 1973; Buechley, 1975; Schimmel and Murawski, 1975, 1976; and Schimmel 1978). However, most of these studies mainly provide, at best, only limited qualitative evidence bearing on possible mortality relationships to nonepisodic PM and SO_2 air pollution (see Appendix 14A) and have, collectively, yielded apparently confusing and conflicting results.

Glasser and Greenburg (1971), for example, reported results from regression and other types of statistical analyses of nonepisodic PM and SO_2 air pollution relationships to New York City mortality, based on the evaluation of both excess mortality deviations from 15-day moving averages during October-March of the years 1960-64 and deviations from "normal" 5-year mortality levels for the same months. Significant relationships between excess mortality and both SO_2 and PM (CoHs) were reported (with the association being stronger for SO_2) based mainly on analysis of unadjusted excess mortality deviations from the 5-year normal averages. However, inspection of the reported data broken down by individual years or by other factors (e.g., day of week, month, etc.) reveal numerous internal inconsistencies in the data, including a number of "reversals" in terms of markedly lower positive (or even negative) mortality deviations at higher PM or SO_2 levels than positive deviations seen at lower pollutant levels. Only in 1963 did there appear to be clear and consistent dose-response relationships, but the pollutant levels during that year were badly confounded with the occurrence of extremely cold temperatures and a major influenza epidemic. About the only other consistent dose-response relationship readily apparent in the overall 5-year data was an apparent steady increase in mortality levels as a function of jointly ascending PM and SO_2 values, suggesting possible interactive effects of the two pollutants on mortality. However, the mortality deviations by PM and SO_2 concentrations were not cross-tabulated for individual years; it was thusly impossible to evaluate the internal consistency of such a possible interactive effect on a yearly basis.

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[a] The results of recent, unpublished analyses (on a year-by-year basis) of associations between daily London mortality rates and BS or SO_2 levels for each winter from 1958-59 to 1971-72 are summarized in Appendix 14F. Those analyses, taking temperature, humidity and temporal trends into account, indicate that significant ($P<.05$) positive associations between London mortality rates and daily SO_2 levels occurred during 1958-59 and 1962-63 as apparently exceptional winters with simultaneously high SO_2 and frequent fog days. However, significant associations between mortality and BS levels were found for winters during additional years, including some such as 1967-68 when 24-hour BS levels exceeded 250 $\mu g/m^3$ only once during the entire winter.

In another study of New York City mortality data in the 1960s, Hodgson (1970) employed multiple regression analyses to examine the relationship between deaths and air pollution on a monthly basis and reported significant correlations between excess deaths and both SO_2 and PM (CoHs). In addition, Lebowitz (1973a,b), utilizing a stimulus-response model to study daily air pollution exposures, meteorology, and mortality in New York (1962-1965), reported that adverse temperature and humidity changes were found to be important but did not completely account for all apparent air pollution-mortality associations. Schimmel and Greenburg (1972) also reported significant associations between excess mortality and nonepisodic air pollution (indexed by PM and SO_2 levels) in New York City during 1963-68, with PM measured as CoHs apparently being more strongly implicated than SO_2.

None of the above studies, regardless of their specific results, permitted significant increases in mortality levels to be quantitatively related to specific PM or SO_2 levels in New York City. This is due, in part, to the particular statistical analyses employed. It is also due to the fact that a single central monitoring station was typically used as an index of variations in PM and SO_2 levels for the entire New York City area, a questionnable practice, especially in view of a report by Goldstein and Landowitz (1977a,b) indicating relatively low ($r < 0.40$) pair-wise correlations between either SO_2 or CoHs readings from that station and comparable readings from other New York City monitoring stations. The lack of such correlation raises serious questions about the probable validity of even the qualitative associations between city-wide mortality and nonepisodic PM or SO_2 air pollution in New York based on the above study results (the overall lack of correlation, in contrast, may be less important during episodic events when very high air pollution (PM and SO_2) levels may have approached a common "ceiling" throughout the city). Additional questions can be raised in view of the large number of correlation coefficients often calculated in the different studies and the probability of some apparently significant associations being found by chance alone.

Besides the above caveats, it should be noted that Buechley et al. (1973), Buechley (1975), Schimmel and Murawski (1975, 1976), and Schimmel (1978) have published further analyses of New York City mortality data, including data from additional years extending into the 1970s, with interesting results in comparison to those of the above earlier studies. For example, Buechley et al. (1973, 1977) evaluated the relationship of daily deaths in the New York/New Jersey metropolitan area (1962 through 1972) to SO_2 measured at a single New York City monitoring station and reported some statistically significant associations, suggesting a relationship between residual mortality and SO_2 levels. However, Beuchley (1975) noted that mortality rates did not decrease in association with a three-fold decline in SO_2 over the 10-year period studied and questioned whether SO_2 was perhaps only serving as a covarying surrogate of other pollutants more directly linked to health effects. In an independent analysis, Schimmel and Murawski (1975, 1976), controlling for common seasonal trends in mortality, temperature, and pollution in New York City found that the central Manhattan monitoring station PM (CoHs) levels varied little over three time periods (1963-66; 1967-69; 1970-72) analyzed, while average SO_2 concentrations declined markedly over that timespan. Based on their analyses, the percentage of premature deaths due to air pollution in the respective periods was estimated to be 2.78, 2.48, and 3.20, i.e., lower percentages attributed to air pollution than those (~4.0 percent) suggested by the earlier analyses of Schimmel and Greenburg, 1972. Further, the percentage attributed to SO_2 was 0.58, 1.22, and 0.62 for the three different time periods (percentages not significantly different from zero), whereas the remaining 80 percent or so appeared to be attributable to PM (CoHs). Also, because percentages of excess deaths attributable to SO_2 varied very little over the three time periods as 3-fold decreases in SO_2 occurred, it was concluded that SO_2 was likely a surrogate for other (possibly unmeasured) variables but not directly related to health effects (mortality) across the range of ambient concentrations (annual means $\leq$0.2 ppm; daily peaks up to 0.4-0.6 ppm) studied. Subsequently, time-series analyses by Schimmel (1978), eliminating seasonal and other cyclical effects on New York City mortality, demonstrated that the regression of mortality on SO_2 was not significant, negative correlations being obtained at times. Overall, then, these later analyses by Schimmel and Murawski (1976) and Schimmel (1978) create serious doubt regarding reported associations between mortality and nonepisodic SO_2 levels present in New York City during the 1963 to 1972 period. The same analyses, however, were interpreted by the authors as being indicative of weak but positive associations between nonepisodic mortality and PM (annual means ≅2.0 to 2.5 CoHs; daily peaks up to 5.0 to 6.0 CoHs) levels in New York City during the same time period. Unfortunately, the use of aerometric data from the single central Manhattan monitoring station as an estimate of pollutant exposures for the entire New York area precludes clear quantitative statements regarding possible effect or no-effect levels based on these results.

14.3.1.3 Morbidity--Studies of morbidity effects associated with acute or short-term air pollution exposures are much less common in the epidemiological literature than morbidity studies of chronic or long-term air pollution exposures. This reflects dual complications of the difficulty of having adequate estimates of individual's pollution exposures as well as the statistical analytical problems of the health data being collected. The main focus here is on studies providing information on quantitative relationships or associations between ambient air concentrations of SO_2 or PM and acute exposure morbidity health effects.

Several British studies have been published on health effects associated with acute or short-term exposures of adults to sulfur oxides and particulate matter which appear to provide useful information on quantitative dose-effect relationships. Illness data were obtained in many of the early severe pollution episodes discussed above; much of this morbidity information did little more than support the mortality results reported in those studies in providing evidence that increases in illness occurred along with increases in deaths.

Certain other studies, however, provide valuable information useful in estimating quantitative relationships between morbidity effects and acute exposures to elevated BS and SO_2 levels in London during the 1950s and 1960s. For example, Waller and Lawther (1955) and Lawther (1958) studied associations between daily variations in smoke and SO_2 pollution and self-indicated health status in 29 British chronic bronchitis patients during the winters of 1954-55 and 1955-56. Patients maintained diaries on which their daily condition was indicated in relation to their usual condition. The alternatives were "better," "same," "worse," and "much worse." During the month of January 1955, an episode of relatively high pollution resulted in a sharp increase in the number of patients whose condition worsened as 24-hour smoke (BS) increased from about 400 $\mu g/m^3$ to 2000 $\mu g/m^3$ and 24-hour SO_2 levels increased from about 450 $\mu g/m^3$ (0.15 ppm) to about 1300 $\mu g/m^3$ (0.50 ppm). Waller and Lawther (1955) reported that, on the worst day of the episode (January 19), when smoke (BS) concentrations in London increased tenfold during the course of 2 hours, there was a deterioration in the clinical condition of some patients with bronchitis. On this day, peak hourly smoke (BS) concentration may have reached 6500 $\mu g/m^3$. Sulfur dioxide also increased to an hourly maximum of about 2860 $\mu g/m^3$ (1.0 ppm) but H_2SO_4 apparently did not, on the basis of washings from impactor slides. Most of the mass of PM was determined by microscopic studies to consist of particles less than 1 μm in diameter (Waller, 1963).

In the winter of 1955-56, the study was extended to include 180 patients in the London area and "degree of illness" scores were quantified as follows: better = -1; same = 0; worse = 1; much worse = 2. Lawther (1958) reported that the exacerbation of preexisting illness appeared to be related more closely to pollution than to temperature or humidity during the 1955 winter months, but the relationship disappeared when the levels of pollution decreased in the spring. Actual numerical data are not tabulated in the Lawther (1958) report, but inspection of the published graphed results indicate that exacerbation of illness notably increased when winter smoke (BS) levels exceeded about 300-350 $\mu g/m^3$ and SO_2 levels about 500-600 $\mu g/m^3$. However, in the spring of the year, pollution concentrations were no longer as clearly associated with health status. That is, the association between pollution and illness decreased when spring smoke (BS) levels fell to a fairly consistent 24-hour concentration of less than 250 $\mu g/m^3$ and the few higher peaks in 24-hour smoke (BS) seemed to have little effect on illness status. Also, although SO_2 reached some intermittent peak concentrations as high as those associated with increased illness during the winter, no marked exacerbation of preexisting illness seemed to occur during the spring. Lawther (1958) noted that the hazard to subjects' health may be better indicated by markedly higher short-term hourly peaks in pollution rather than the 24-hour average levels noted above and that the results are not necessarily indicative of causal relationships, but rather that the measurements of smoke (BS) and SO_2 may only be indicators of some other causal agent.

A later report by Lawther et al. (1970) both reviewed the 1954-55 and 1955-56 results and, also, provided results for further extension of these studies into the winters of

1959-60, 1964-65, and 1967-68. The techniques used in the later studies were similar to the earlier ones except that the patients now reported on health status in relation to the previous day rather than in relation to usual conditions. These studies confirmed results from previous years in that the worsening of health status continued to be associated with increases in air pollution. The responses of 1,071 patients were evaluated for the winter of 1959-60 in relation to variations in BS and SO_2 levels measured at the same seven London air monitoring sites as those yielding aerometric data used by Martin and Bradley (1960) and Martin (1964) to evaluate daily variations in London mortality. Similarly, responses from 1,037 patients were evaluated for 1964-65 in relation to pollution levels measured at the same sites. Although there were fewer days of high pollution in 1964-65 than in 1959-60, and an impression of a slightly reduced and less consistent response in 1964-65 compared to 1959-60, clear positive associations still persisted between worsening of bronchitic patients' conditions and the air pollution variables measured in 1964-65. Further, Lawther et al. (1970) stated that, although exact relationships between the responses of patients and the concentrations of smoke and SO_2 could not be determined, the minimum pollution leading to any significant response was about 500 $\mu g/m^3$ (0.17 ppm) SO_2, together with about 250 $\mu g/m^3$ smoke (BS). Lawther et al. (1970) also speculated that these responses may reflect the effects of brief exposures to maximum concentrations several times greater than the 24-hour average but provided no data analyses clearly substantiating this hypothesis. As in the earlier studies, the exacerbation or worsening of health status appeared to relate more closely to pollution indices during the first part of the winter, and in some instances there was little reponse to higher concentrations of pollutants near the end of the winter. Although the concentrations of smoke and SO_2 closely correlate, examination of the data again suggests that often higher concentrations of SO_2 near the end of the winter, occurring with generally lower concentrations of smoke, produced less response in the study subjects than did the same concentrations of SO_2 earlier in the winter, when smoke was higher. However, there was some evidence of a loss of interest by participants over time, which may also partially explain this pattern.

Follow-up studies compared responses for a selected group of patients (apparently among the most sensitive to BS and SO_2 pollution) for 1964-65 and 1967-68 winters, as shown in Table 14-2. Examination of the data reported for the winters of 1964-65 and 1967-68 for the group of selected patients shows statistically significant associations of morbidity effects with pollution for both winters. This includes the winter of 1967-68 when, as Lawther et al. (1970) noted, there were hardly any periods of high pollution. In fact, the 24-hour BS levels, except on one occasion, never exceeded 250 $\mu g/m^3$ and SO_2 levels were consistently below 500 $\mu g/m^3$. Lawther et al. (1970) thought it likely that these selected patients were especially sensitive to pollution since no significant correlations were expected considering the low pollution levels to which this group was exposed.

These studies among chronic bronchitis patients in London continued into the 1970s as the frequency of periods of high pollution declined. Lawther et al. (1973, 1974a,b,c) reported acute

TABLE 14-2. SUMMARY OF RESULTS, SELECTED PATIENTS, 1964-65 AND 1967-68

		1964-65	1967-68
Mean Score	Mean	1.98	1.96
	S.D.	0.10	0.11
Smoke (μg/m^3	Mean	129	68
	S.D.	95	48
SO_2 (μg/m^3)	Mean	264	204
	S.D.	149	100
H_2SO_4 (μg/m^3)	Mean	7.3	6.3
	S.D.	4.8	4.0
Temp (°C)	Mean	6.4	6.3
	S.D.	4.1	4.3
Corr. coeff., mean score and	Smoke	0.39*	0.31*
	SO_2	0.30*	0.28*
	H_2SO_4	0.51*	0.26*
	Temp.	0.24*	0.17*

These results are for the whole winter period. October to March.

*Significant at .05 level.

Source: Lawther et al. 1970.

decrements in ventilatory function in four healthy adults and 2 bronchitics in London to be associated with variations in air pollutant levels measured at their place of work or treatment. After multiple regression analysis to remove time trend effects, SO_2 concentrations explained the largest proportion of residual variance in peak flowrates, with clearest associations shown after walking exercise in heavy pollution.

Martin (1964) examined applications for hospital admissions in London for the winter of 1958-59 and found for men ages 45 to 79 (after adjustment for day of the week and correction for 15-day moving average) significant correlations for both cardiovascular and respiratory conditions with smoke (r = 0.46) and SO_2 (r = 0.40). Analogous significant correlations were found for the same male age group for such conditions in relation to both smoke (r =0.41) and SO_2 (r = 0.43) for the winter of 1959-60. The average deviations associated with increasing SO_2 and smoke levels during both winters are summarized in Tables 14-3 and 14-4. As seen in those tables, whereas no clear threshold for the onset of mean positive deviations across the exposure ranges can be distinguished, very marked increments in the positive mean deviations can be discerned starting at 800-899 $\mu g/m^3$ for SO_2 and 1100-1510 $\mu g/m^3$ for PM (BS). Presentation by the authors of their results separately in relation to BS and SO_2 (and the present summarization in Tables 14-3 and 14-4) is not meant to imply that the relative individual contributions of BS and SO_2 alone to the observed effects can be ascribed to the concentrations listed, in view of considerable covariation in SO_2 and BS levels during the two winters studied.

Essentially no other British or European epidemiological studies on morbidity effects associated with acute exposures to PM or SO_2 appear to yield pertinent data useful in attempting to quantify such relationships for present criteria development purposes. Few even provide good qualitative evidence characterizing acute PM and SO_2 exposure-morbidity relationships. See Appendix 14A for brief annotated comments regarding such additional studies evaluated by the present authors.

In addition to the quantitative British studies reviewed above, several American studies appear to provide limited but useful qualitative evidence for the association of particular types of morbidity effects with acute (24 hour) exposures to SO_2 and PM. For example, Greenburg et al. (1962, 1963) reported that during the 1953 New York City air pollution episode, statistically significant increases in emergency clinic visits for upper respiratory infections and cardiac illnesses, respectively, occurred at 3 of 4 and 2 of 4 New York City hospitals studied, but no significant increases occurred in asthma clinic visits. During a 1962 New York episode, however, no significant increases in hospital clinic visits were detected (Greenburg et al., 1963) for respiratory infections or cardiac illnesses, although significant increases in physician visits to treat respiratory complaints among elderly residents at 4 old-age homes were observed. Greenburg et al., (1967) further reported significant increases in emergency clinic visits for bronchitis and asthma at 3 of 7 New York City hospitals on the third day of the Thanksgiving Day weekend episode in New York in 1966.

TABLE 14-3. AVERAGE DEVIATION OF RESPIRATORY AND CARDIAC MORBIDITY FROM 15-DAY MOVING AVERAGE, BY SMOKE LEVEL (BS) (LONDON, 1958-60)

Smoke level ($\mu g/m^3$, BS)	Number of days	Mean deviation
500-599	9	3.2
600-699	6	-0.7
700-799	9	2.4
800-1099	8	4.9
1100-1510	7	12.9

Source: Martin (1964).

TABLE 14-4. AVERAGE DEVIATION OF RESPIRATORY AND CARDIAC MORBIDITY FROM 15-DAY MOVING AVERAGE, BY SO_2 LEVEL (LONDON, 1958-60)

SO_2 level ($\mu g/m^3$)	Number of days	Mean deviation
400-499	9	2.2
500-599	6	5.1
600-799	9	6.9
800-899	6	12.8
900-1280	5	12.8

Source: Martin (1964).

However, these reported increases in numbers of daily emergency visits may have in part been due to the clinics having been closed during the preceding holiday weekend. Overall, these results suggest that increased cardiac and upper respiratory tract morbidity may have significantly increased, especially among the elderly, during episodic marked elevations of PM (5.0-8.0 CoHs) or SO_2 (0.4-0.5 ppm; 1050-1310 $\mu g/m^3$). However, the one isolated instance (November 1966) of increased asthma visits being associated even with the episodic elevations can be reasonably questioned in light of other observations by Greenburg et al. (1964) that demonstrate regular seasonal increases in asthma attacks in New York in the fall months (even in the absence of notable peaks in air pollution) and link increases in such attacks to rapid shifts in climatic conditions from warm to cold temperatures.

In other studies focusing on possible morbidity effects associated with nonepisodic exposures to air pollution, McCarroll (McCarroll and Bradley, 1966; McCarroll et al., 1966) and colleagues (Mountain et al., 1968; Thompson et al. 1970; Cassell et al., 1969, 1972; Lebowitz et al., 1972) studied frequency of cough and the "common cold" in New York City residents in relation to SO_2 and PM measurements obtained from monitoring stations within 2000 feet of their apartment building residences. These studies demonstrated significant multiple correlations between acute respiratory symptoms and air pollution, controlling for season, weather, and social class, when seasonal SO_2 means were in the range of 0.10-0.24 ppm (~280-700 $\mu g/m^3$) and seasonal smoke shade means were in the range of 1.56-3.15 CoHs. Initially, multiple regression analyses showed conflicting findings in that the pollutants were occasionally absent from or negative in their regressions. This led to a separation of the combined meteorological and air polluant conditions into categories of: (1) stormy weather (low temperatures, occasional precipitation, high wind speed) when the pollutants were low; (2) stagnation periods (low wind speed, moderate temperatures) when SO_2 and TSP were high; (3) periods of change in pollutant levels during the fall through spring periods; and (4) high photoxidant conditions in the summer. This analysis yielded significant correlations between the pollutants and acute symptoms for 1800 individuals studied weekly in New York (1962-65) during stagnation periods and significant correlations of the same acute respiratory symptoms (predominately common colds) with meteorological conditions during stormy periods. They also found a lag of one to three days in symptoms and corresponding increases in school absenteeism. Some individuals (mostly under the age of 10) were found to have reacted consistently and frequently to increases in the pollutants, and their respiratory symptoms were of greater duration and severity than "nonsensitive" individuals (Lebowitz et al., 1972). Those who were "sensitive" during the first part of the period under study were found to be "sensitive" later on in the study. The attack rates per person year were about double for the "sensitive", and occurred predominantly in the winter period. However, while suggesting that respiratory symptom morbidity effects may be qualitatively associated (especially for some "sensitive" children) with nonepisodic increases in air pollution these studies do not allow specific levels of PM or SO_2 or their individual contributions to be related to the observed effects.

Further studies have attempted to relate increased morbidity effects to acut (24 hour) increases in air pollution containing PM and SO_2 occurring in American cities during the late 1960s or 1970s. Only a few, however, were conducted methodologically in such a manner so as to have employed community PM or SO_2 aerometric data that can be viewed as being somewhat reasonably representative of ambient outdoor exposures of the particular populations studied. Some of the more important findings from those studies having potentially important bearing on present criteria development efforts are evaluated next (see Appendix 14D for comments on other, less useful studies for present criteria development purposes).

Some studies have examined relationships between acute elevations in PM or SO_2 air pollution in certain American cities and the occurrence of asthma attacks, yielding at best equivocal results which, taken together, tend to call into question whether any such relationships existed at ambient air PM or SO_2 levels present in the cities studied. For example, Cohen et al. (1972) reported a weak association to exist between air pollution and the frequency of asthma attacks in a preliminary pilot study of 43 self-reported asthmatics living near a coal-fueled power plant in West Virginia. Temperature was reported to be most strongly correlated with the frequency of attacks based on limited data analyzed for 20 subjects having at least one attack during the study (note that several subjects also dropped out of the study or returned health survey data too infrequently during the study to be included in data analysis). Based on multiple regression analysis of the remaining data, both SO_2 and TSP were reported to be significantly ($P<.01$) correlated with attack rate after adjusting for temperature; also, attack rates were reported to be significantly higher on high air pollution days (TSP $\geq$ 150 $\mu g/m^3$; $SO_2 \geq$ 200 $\mu g/m^3$) defined by aerometric data monitored within 1.5 mile of the subjects' residences. However, these reported findings cannot be accepted as validly indicating that asthmatic attacks were associated with the ambient PM or SO_2 concentrations present at the time. This is due to: (1) the reported large subject dropout rate and deletion of data due to too-infrequent subject reports of health status; (2) the likely resulting bias introduced into the data ultimately analyzed, along with any biasing of results due to deletion of data for subjects reporting no attacks; (3) the lack of information on changes in panel membership over time; and (4) insufficient information about the specific data analyses employed to assure that proper adjustments for seasonal or other time-related factors were carried out.

In addition to the above caveats regarding the Cohen study, it should be noted that Goldstein and Dulberg (1981) reported finding no significant relationships between asthma "events" (i.e., increased hospital emergency room visits for asthma attacks) and acute 24 hr changes in PM or SO_2 air pollution levels in New York City during 1969-71 (when average daily PM levels $\cong$0.9-1.5 CoHs and mean daily SO_2 $\cong$0.03-0.07 ppm or 80-180 $\mu g/m^3$) as measured by a 40-station sampling network. Unfortunately, the authors did not adjust for temperature change (as a key variable affecting asthma attack occurrence), thus making it difficult to fully accept their reported "no-effect" findings until more proper analyses including temperature adjustments are carried out. However, in another study that did adjust for temperature and other important potentially confounding factors, Samet et al., (1981) found no increases in hospital visits for all respiratory diseases (including asthma) in Stuebenville, Ohio, when 24-hr means for TSP fell in the range of 14-696 $\mu g/m^3$ and 24 hr SO_2 levels were 2-369 $\mu g/m^3$, as monitored at a centrally located station proximal (within 300 meters) to the hospital studied. Many residences of potentially affected subjects, however, were located further away from the monitoring site, making it unclear as to what extent the stated TSP and SO_2 values can be taken to represent "no-observed effect" levels. Nevertheless, the weight of all of the above considerations still points towards the conclusion that the studies reviewed here provide essentially no evidence for asthma attacks being associated with acute exposures to low ambient 24-hr PM or SO_2 levels encountered in several different American cities in the late 1960s or 1970s.

The results reported by Samet et al., (1981) concern some of the findings just beginning to emerge from a long-term epidemiological study of morbidity-air pollution relationships in several American cities. As part of this so-called "Six-Cities Study," Samet et al., (1981) evaluated additional indices of respiratory effects, besides those noted above for asthma attacks, in heavily industrialized Steubenville, Ohio. Hospital emergency room visits for various respiratory diseases during March, April, October, and November of 1974-1977 were related to daily levels of TSP, SO_2, NO_2, CO, and O_3. After adjusting for meteorological variables, weekly, seasonal, and yearly cycles, possible day-of-week effects, and multicollinearity of the pollutants, deviations from average numbers of daily emergency visits for all respiratory diseases did not vary significantly with pollution, although the largest deviations occurred at the highest pollutant levels (see ranges stated above) for TSP and SO_2. For respiratory diseases, a linear regression model identified a significant effect of both unlagged TSP and SO_2, but not NO_2. However, the authors warn that this result should be viewed with caution since, although the regression coefficients attained significance at $P<.05$, the contribution of pollutant variables to the multiple R^2 is only 0.01 - that is, only 1 percent of the variance of the respiratory disease index is explained by TSP or SO_2. Furthermore, the days of highest TSP ($\geq 202\ \mu g/m^3$) or SO_2 ($\geq 121\ \mu g/m^3$) pollution were not significantly associated with increased numbers of emergency room visits for respiratory treatments. Other (more ambiguous) preliminary results from the Six-City Study, as reported by Dockery et al., (1981), are commented on in Appendix 14A.

In summary, the above studies on acute exposure effects tend to suggest that the elderly, those with chronic cardiorespiratory diseases, and children may constitute populations at risk for manifesting morbidity effects in response to acute exposure to elevated atmospheric levels of sulfur dioxide and particulate matter. Qualitatively, increases in the occurrence of cardiac and upper respiratory tract disease symptoms, including exacerberation of preexisting chronic bronchitis (but not asthma attacks) appear to be among the morbidity effects most clearly associated with exposures to the ambient levels of PM and SO_2 evaluated in the studies assessed above and are most clearly seen at markedly elevated levels of the two pollutants. For example, increased applications by adults aged 45-79 for admissions to London hospitals for cardiac and respiratory morbidity most clearly occurred, based on the Martin (1964) study, when 24-hour BS and SO_2 levels approached or exceeded 900-1000 $\mu g/m^3$; but Martin's data also suggest that such effects may have occurred at somewhat lower levels down to 500 $\mu g/m^3$ for both SO_2 and BS. Using a probably more sensitive morbidity indicator, Lawther's studies in London further appear to demonstrate that worsening of health status among bronchitic patients is associated with acute 24-hour exposures to BS of 250-500 $\mu g/m^3$ in the presence of SO_2 levels in the range of 500-600 $\mu g/m^3$. In contrast, no effects on most bronchitics appeared to be detectable at 24-hour BS levels below 250 $\mu g/m^3$ in the presence of 24-hour SO_2 levels below 500 $\mu g/m^3$; limited evidence suggests some effects on a selected group of highly sensitive bronchitics at somewhat lower, but not precisely-defined BS and SO_2 levels. Similarly, American studies by Greenburg's group appear to most clearly demonstrate increased cardiac and upper respiratory morbidity, especially among the elderly, during air pollution episodes in New York City when extremely high levels of PM (5.0-8.0 CoHs) and SO_2 (>1000 $\mu g/m^3$) were present. On the other hand, much less clearly demonstrated were morbidity effects related to nonepisodic elevations in air pollution containing PM and SO_2. The findings of McCarroll's group (especially as reported by Lebowitz et al., 1972) in New York City, for example, suggest at most an increase in upper respiratory tract symptoms (e.g., coughs and colds) in certain "sensitive" children at lower nonepisodic levels of PM or SO_2. Insufficient epidemiological information from the studies of McCarroll and coworkers exists, however, by which to determine specific quantitative acute exposure levels at which the health of such "sensitive" children might be adversely affected.

14.4 CHRONIC PM/SO_2 EXPOSURE EFFECTS

14.4.1 Mortality

Numerous studies have been performed to compare general or cause-specific mortality in areas of lowest-to-highest pollution concentrations. However, virtually all of these studies: (1) used aerometric data of questionable accuracy or representativeness of study population exposures; and (2) did not adequately account for the potential effects on mortality rates of such confounding factors as cigarette smoking, occupation, social status, or mobility differences between areas (see Appendix 14A). These methodological problems preclude accurate characterization of any quantitative relationships between mortality and air pollution parameters. Therefore, essentially no epidemiological studies are presently well-accepted as providing valid quantitative data relating respiratory disease or other types of mortality to chronic (annual average) exposures to sulfur oxides or particulate matter. On the other hand, the findings of certain published studies of chronic air pollution effects on mortality may warrant assessment here in regard to their potential for establishing qualitative links between mortality and chronic exposures to PM or sulfur oxides. Two types of general approaches have been employed in such studies: (1) aggregation of mortality and other information, e.g. smoking, or socioeconomic status data, in relation to specific individuals within the study population(s); and (2) aggregation of analogous data for entire populations across large geographic areas, e.g. cities, counties, or standard metropolitan statistical areas (SMSAs). Examples of results obtained with each type of approach are evaluated below.

Among the best known and most often cited examples of the first approach listed above are the Winkelstein et al. (1967), Winkelstein and Kantor (1967), and Winkelstein and Gay (1971) studies of total and cause-specific mortality in Buffalo and Erie County, New York, during 1959 to 1961. A network of 21 sampling stations provided data on TSP (hi-vol sampler) and oxides of sulfur (non-specific sulfation methods) for the period July 1961 to June 1963; and these aerometric data were used to categorize geographic areas as "low" to "high" air pollution areas. Chronic respiratory disease mortality for white males 50 to 69 years old was reported to be about three times higher in the high-pollution areas than in the low-pollution areas, across all economic groups (Winkelstein et al. 1967). Additional positive associations in relation to TSP concentrations were reported for both stomach cancer (Winkelstein and Kantor, 1967) and deaths from cirrhosis of the liver (Winkelstein and Gay, 1971).

However, numerous criticisms of these studies can be noted which raise serious doubts regarding the validity of the reported findings, including the following most salient methodological problems: (1) the use of 1961-1963 TSP and sulfur oxides measurement data as a basis for retrospectively classifying geographic areas according to presumed past air pollution gradients against which to compare mortality among the elderly during 1959-1961; (2) inadequate controls for possible age differences between study groups that may have covaried with the air pollution gradient used; (3) lack of information on lifetime (including occupational) exposures to PM or SO_2; (4) failure to correct for smoking habits; and (5) the implausibility of some of the reported findings, e.g., air pollution increasing mortality due to liver cirrhosis. In a later presentation, Winkelstein (1972) commented on several of these points and attempted to correct for some of them, such as by looking at smoking patterns among certain populations living in some of the same study areas included in the earlier analyses. Still, the 1972 analyses do not adequately counter the different major concerns regarding the Winkelstein Buffalo mortality findings. For example, the later finding in the follow-up investigation (Winkelstein, 1972) of no significant differences in smoking patterns among the different study areas for females does not adequately control for possible smoking effects in different specific population cohorts evaluated in the earlier study analyses reported by Winkelstein and coworkers. These particular studies, therefore, are of questionable validity even in regard to providing credible qualitative evidence for links between PM air pollution and mortality in Buffalo during 1959-1961.

Turning to examples of the second type of approach listed above as being used for evaluating chronic air pollution effects on mortality, Lave and Seskin (1970) employed regression analyses to evaluate relationships between mortality rates and indices of air pollution in Britain such as PM levels (measured by deposit gauges and BS readings) and SO_2 levels, all as reported by Stocks (1959, 1960) and Ashley (1967). Significant positive associations were reported to exist between PM levels (as indicated by deposit gauge and BS measurements) taking into account socioeconomic status (SES). This Lave and Seskin (1970) study has been criticized (Holland et al., 1979) regarding, for example, difficulties in justifying inclusion of SES and air pollution levels in the regression analyses as if they were completely independent variables and the failure to make some direct allowance for smoking habits in the analyses. However, perhaps even more basic difficulties with the analyses derive from: (1) the use of imprecise qualitative estimates of PM pollution (e.g., deposit gauge data and BS aerometric data expressed in terms of mass concentration estimates not appropriately obtained by means of site-specific calibrations of reflectance readings against local gravimetric measurement data); and (2) ambiguities regarding locations of sampling devices in relation to study populations, therefore raising questions about the representativeness of the aerometric data in estimating PM exposure levels.

Other studies discussed by Lave and Seskin (1970) and in three further publications (Lave and Seskin 1972, 1977; Chappie and Lave, 1981) extended the original U.K. analysis approach (Lave and Seskin, 1970) to standard metropolitan statistical areas (SMSAs) in the United States. Based on such analyses, analogous positive associations between mortality and air pollution variables were reported for the United States. Many criticisms similar to those indicated above for the earlier Lave and Seskin (1970) publication apply here. Of crucial importance are basic difficulties associated with all of their analyses in terms of: (1) use of aerometric data without regard to quality assurance considerations, most notably including use of sulfate measurements known to be of questionable accuracy due to artifact formation during air sampling (see Chapter 3); and (2) questions regarding the representativeness of the air pollution data used in the analyses as estimates of actual exposures of individuals included in their study populations. For example, in some instances data from a single monitoring station were apparently used to estimate pollutant exposures for study populations from surrounding large metropolitan areas. Clearly, then, no useful information on quantitative relationships between specific concentrations of PM or sulfur oxides and mortality can be derived from these published analyses. However, the question remains as to whether any clearly consistent qualitative conclusions regarding PM or SO_x air pollution-mortality relationships can be drawn based on these and analogous studies, as discussed next.

Lave and Seskin (1972, 1977) reported first on the results of linear regression analyses of relationships between: (1) mortality (mainly total mortality rates; TMR) in more than 100 United States SMSAs; (2) various measures of air pollution, such as sulfur dioxide (SO_2); suspended sulfates; total suspended particulate matter; and others (CO, NO_2, ozone, hydrocarbons, etc.); and (3) variables serving as proxies for certain demographic factors such as age, sex, race, population density, and socioeconomic status known to be important determinants of total and cause-specific mortality rates. Using such indices, aggregated by geographic areas, Lave and Seskin (1977) reported results of regression analyses which they interpreted as demonstrating consistent significant positive associations between air pollution in general or particular pollutants (e.g., sulfates or TSP) and TMR recorded in 1960 for 117 U.S. SMSAs. They went on to state that the implications of their findings meant, for example, that for every 10 percent change (increase or decrease) in sulfate or TSP air pollution levels, there would be a resulting change in TMR of 0.50 percent (for sulfates) and 0.44 percent (for TSP); in other words, decreases of 4.60 and 3.99 in deaths per 100,000 could be expected for every 10 percent reduction in ambient air sulfates or TSP concentrations, respectively.

Lave and Seskin (1977) reported further results obtained with similar regression analyses of mortality, air pollution and demographic data for additional years (1961-69) for most of the same SMSAs. This included presentation of comparisons among results obtained with: (1) analyses of data for all SMSAs or SMSA subsets for different years, e.g. results for 1960 versus 1969, (2) analysis of data for different individual SMSAs for the same year(s), e.g., data for Chicago versus St. Louis, Philadelphia, and certain other SMSAs for 1962-69; and (3) use of various models beside the basic linear regression model used for earlier analyses. Chappie and Lave (1981) have since reported additional analyses for previously unavailable 1974 data for 104 SMSAs. This latter paper compared (1) results obtained with the linear model for 1974 data versus earlier (1960, 1969) results; (2) results obtained for 1974 data with various models in addition to the basic Lave and Seskin linear model; and (3) results obtained for the 1974 data as a consequence of adding or deleting various potential explanatory variables or proxy variables from the models. Table 14-5 summarizes comparison of some salient findings from the 1960, 1969, and 1974 analyses that led to key conclusions stated by Chappie and Lave (1981). Those conclusions include: (1) that a strong, consistent and statistically significant association between sulfates and mortality persists across all years studied (and that the association is little changed by adding variables for smoking and alcohol consumption, by using city or county instead of SMSA data or by adding a medical care variable and going to a simultaneous equation framework); (2) that the association between particulate matter and mortality, while previously consistently positive and significant in 1960 and 1969, was no longer significant or consistently positive in 1974; and (3) that these results support and strengthen the earlier conclusions of Lave and Seskin (1977) that "stringent abatement of sulfur oxides and particulates would produce social benefits (based on health effects alone) that greatly exceed social costs."

Lave and Seskin (1977) stated the above conclusions and implications despite: (1) their own expressed misgivings concerning the quality, accuracy and representativeness of the aerometric data used as population exposure estimates and other data used as inputs to serve as proxies for key parameters or components of their regression model(s); (2) ambiguities and concerns, again recognized by the authors themselves, regarding the appropriateness of the main linear (OLS) model they used to represent the underlying relationships they were attempting to model; and (3) numerous internal inconsistencies in their reported results which do not support or which contradict their main conclusions. A few examples illustrating the third type of problem will suffice here.

TABLE 14-5. SUMMARY OF KEY RESULTS REGARDING MORTALITY-AIR POLLUTION RELATIONSHIPS IN UNITED STATES CITIES BASED ON LAVE AND SESKIN MODEL ANALYSES FOR 1960, 1969, AND 1974 DATA

Dependent variables	1960 TMR[a]	1969 TMR[b]	1974 TMR[c]
Air Pollution Variables:			
Min S	4.733	-0.384	0.294
	1.67	-0.07	0.04
Mean S	1.726	6.329	16.915
	0.53	1.81	3.09
Max S	0.279	-0.527	-1.809
	0.25	-0.67	-2.09
Sum S Elas =	0.50	0.59	1.32
Min P	0.199	0.434	2.366
	0.32	0.69	1.32
Mean P	0.303	0.055	-1.386
	0.71	0.13	-1.51
Max P	-0.018	0.130	0.294
	-0.19	1.83	1.80
Sum P Elas =	0.44	0.56	0.06
Socioeconomic Variables:			
% ≥65	68.802	64.030	64.265
	16.63	17.11	17.59
% Nonwhite	3.960	2.037	2.000
	3.82	2.24	1.96
% Poor	0.384	5.113	5.148
	0.26	2.13	2.15
Density	8.285	0.013	9.687
	1.54	2.51	1.92
Log Popn	-27.566	-42.774	-44.594
	-1.38	-2.22	-2.80
Other Terms:			
R^2	0.831	0.817	0.861
N	117	112	104
Constant	343.230	387.011	313.342
	2.87	3.36	3.19

[a] Regression (1960 TMR) is from Lave and Seskin (1977; p. 31 - Regression 3.1-1).

[b] Regression (1969 TMR) is from Lave and Seskin (1977; p. 121 - Regression 7.1-3).

[c] Regression (1974 TMR) is from Chappie and Lave (1981; Table 1 - Regression 1-1).

Abbreviations: TMR=Total Mortality Rates; Min S, Mean S, Max S = average minimum sulfate levels, etc.; Min P, Mean P, Max P = average minimum TSP levels, and so on. Numbers beneath the regression coefficients for dependent variables are t-statistics. The sums of elasticities represent the estimated percent change in the dependent variable resulting from a ten percent change in the three air pollution variables.

For instance, in discussing the 1960 results shown in Table 14-5, Lave and Seskin (1977) noted not only that "the results are encouraging in that more than 83 percent of the total mortality rate across the 117 SMSAs was accounted for by the eleven dependent variables", but also that "the estimated coefficients for the air pollution variables were disappointing in that none had a statistically significant coefficient, and one coefficient was negative; however the six variables made a statistically significant contribution as a group." The latter statement is based on an F-test comparison of the regression results obtained without the six pollutant variables present versus with them included in the regression analysis, which rejected strongly the hypothesis that the apparent effects of the air quality variables could be attributed to random sampling variability ($P < 1/350$). Tests of the other two regressions in Table 14-5 rejected that hypothesis even more decisively. Thus the air quality measurements appeared to be related to the inter-SMSA differentials in mortality rates, but in view of the non-significance of the t-statistics, the causal source of the significance remained obscured.

In order to clarify the relationship, Lave and Seskin (1970) experimented with various alternative specifications but with indecisive results. As part of such experimentation, they dropped "superfluous variables in order to derive an equation with predicted signs, plausible magnitudes, and statistical significance" and retained only those air pollution variables "whose coefficients were positive and exceeded their standard errors, with the further constraint that at least one sulfate and one particulate measurement were retained." They concluded that: "In reestimating the relationship, we often found that the retained air pollution variables were now significant"; and "sometimes the retained air pollution variable contributed little to the significance of the regression. Such variables were eliminated, subject to the restriction that at least one air pollution variable be retained in the final equation. This technique was used throughout our analyses of other mortality rates."

Several examples can be cited to illustrate problems associated with directly extrapolating data generated by this selective technique to the general population when drawing conclusions about the contribution of air pollution to mortality. For example, Lave and Seskin (1977) reported that analyses of SO_2-mortality relationships in Chicago during 1962-63 showed no significant association between daily mortality and SO_2, using a regression analysis (including SO_2, climatic variables, and day-of-week variables) which explained only 27 percent of the variance, implying that important factors affecting mortality were missing from the regression. Regressing daily deaths lagged against high levels of SO_2 occurring 1, 2, 4 or 5 days prior to the observed deaths yielded some positive associations, but only that for the 5-day lag was significant. Carrying out similar analyses for 1963-64 data produced three of six significant positive associations for SO_2 and daily mortality. However, when similar analyses were carried out for SO_2-daily mortality relationships in Denver and St. Louis, no significant associations were found; nor did any of several other air pollution variables (NO_2, CO, hydrocarbons, etc.) make a significant contribution to analogous analyses for these or other cities (except for Philadelphia; nitric oxide). In fact, the sums of elasticities of the air pollution variables were said to be "generally small and often negative." Lave and Seskin (1977) offered several possible explanations for the differences in results obtained between Chicago and other cities, noting, for example, that Chicago's mean SO_2 levels were almost 4 times as high as those in Denver and 10 times higher than those in St. Louis and that at levels of air pollution substantially below those in Chicago acute effects may not be important. Nevertheless, despite these clear indications against extrapolation to other cities of the Chicago results for SO_2, Lave and Seskin (1977) cite the results for Chicago as if they are generally applicable to all U.S. cities across all levels of air pollution concentrations.

Similarly, Lave and Seskin (1977) appear to make overly broad generalizations concerning: (1) results obtained with their earlier 1960 cross-sectional regression (summarized in Table 14-5) and interpreted as indicating a 50 percent reduction in air pollution (as measured by sulfates and TSP) was associated with a 4.7 percent decrease in unadjusted TMR; (2) results from their cross-sectional analysis of a 1969 subset of SMSA data interpreted as indicating a 50 percent reduction in SO_2 to result in a 2.7 percent decrease in TMR; and (3) results of cross-sectional time-series analysis of a larger, multiple SMSA data set leading to estimations of a 50 percent reduction in SO_2 being associated with a 1.2 percent reduction in TMR. In making the above reported generalizations, Lave and Seskin (1977) do not mention inconsistencies of the above types for the 1962-63 and 1963-64 data analysis results or other numerous inconsistencies and contradictory outcomes present among results from the 1960, 1969, and cross-sectional time-series analyses. Nor did they highlight the fact that their analysis of residuals, as reported by Lave and Seskin (1977), indicated that their regression equation(s) overpredicted unadjusted and age-sex-race adjusted TMR for many of their study SMSAs geographically widely dispersed around the United States. See Table 14-6, which is reproduced from Lave and Seskin (1977) and shows many examples of overprediction of the contribution of air pollution to mortality as indicated by negative residuals for various SMSAs listed.

Numerous other inconsistencies and problems with the Lave and Seskin (1970, 1972, 1977) analyses have been noted by others (Crocker et al. 1979; Lipfert, 1980; Gerking and Schultze, 1981; Ware et al. 1981); and the Chappie and Lave (1981) analyses attempt to correct for some of the more serious criticisms advanced, with some success. However, some of the major improvements made in carrying out certain new analyses reported in the Chappie and Lave (1981) paper have not been employed in reanalysis of the 1960-1969 data earlier reported by Lave and Seskin (1977) and one is left, at this time, with a confusing array of often internally inconsistent and conflicting results derived from the series of analyses reported by Lave and Seskin (1970, 1972, 1977) and Chappie and Lave (1981).

Further difficulties in discerning consistent patterns of association between mortality and air pollution variables are encountered when the results of Lave and coworkers are compared with those obtained by others using analogous "macro-epidemiological" approaches. For example, compare the findings reported by Lave and coworkers of (1) consistently significant

TABLE 14-6. SUMMARY OF LAVE AND SESKIN (1977) ANALYSIS OF RESIDUALS FROM REGRESSION ANALYSES FOR 1960 AND 1969 U.S. SMSA DATA

	Unadjusted total mortality rate		*Age-sex-race-adjusted total mortality rate*	
	1960	*1969*	*1960*	*1969*
Ten largest SMSAs				
New York, N.Y.	−7.08	−26.90	−30.01	−23.29
Chicago, Ill.	35.33	56.76	20.23	45.43
Los Angeles, Calif.	−41.18	−8.16	−35.72	−11.68
Philadelphia, Pa.	11.25	14.29	−12.39	3.96
Boston, Mass.	61.83	15.19	35.07	12.80
Detroit, Mich.	−28.77	9.31	−19.87	9.14
San Francisco, Calif.	31.12	42.74	15.16	32.30
Pittsburgh, Pa.	59.32	—	46.74	—
Saint Louis, Mo.	8.25	−27.30	−13.98	−22.12
Cleveland, Ohio	−29.37	−3.25	−15.72	−17.48
Southwestern SMSAs				
Albuquerque, N.Mex.	−3.30	−32.51	26.15	−49.99
Denver, Colo.	12.95	−82.41	10.29	−76.14
Las Vegas, Nev.	92.12	67.36	54.22	41.66
Los Angeles, Calif.	−41.18	−8.16	−35.72	−11.68
Phoenix, Ariz.	−62.95	−34.38	−70.20	−20.73
Salt Lake City, Utah	−77.14	−90.01	−78.03	−73.52
San Diego, Calif.	−31.20	−89.19	−32.47	−61.65
San Jose, Calif.	−77.74	−53.92	−101.29	−89.74
Ten largest 1960 residuals (unadjusted total mortality rates)				
Tampa, Fla.	−246.09	−202.66	−137.66	−89.13
Wilkes-Barre, Pa.	225.74	175.38	195.06	122.90
Scranton, Pa.	212.09	—	280.72	—
Austin, Tex.	−143.40	—	−159.59	—
Savannah, Ga.	126.41	79.38	112.97	34.40
New Orleans, La.	125.22	78.07	138.52	83.94
Canton, Ohio	−122.61	55.08	−129.57	58.39
Orlando, Fla.	−119.10	—	−107.28	—
Terre Haute, Ind.	113.26	—	79.30	—
Sioux Falls, S.Dak.	−188.00	—	−172.94	—
Ten largest 1960 residuals (age-sex-race-adjusted total mortality rates)				
Scranton, Pa.	212.09	—	280.72	—
Wilkes-Barre, Pa.	225.74	175.38	195.06	122.90
Sioux Falls, S.Dak.	−188.00	—	−172.94	—
Austin, Tex.	−143.40	—	−159.59	—
New Orleans, La.	125.22	78.07	138.52	83.94
Tampa, Fla.	−246.09	−202.66	−137.66	−89.13
Canton, Ohio	−122.63	55.08	−129.57	58.39
Brockton, Mass.	88.87	—	115.18	—
Savannah, Ga.	126.41	79.38	112.97	34.40
Fall River, Mass.	75.84	45.53	108.77	29.77
Ten largest 1969 residuals (unadjusted total mortality rates)				
Duluth, Minn.	62.35	209.72	48.07	161.62
Tampa, Fla.	−246.09	−202.66	−137.66	−89.13
Honolulu, Hawaii	—	−181.21	—	−135.48
Wilkes-Barre, Pa.	225.74	175.38	195.06	−122.90
Fargo, N.Dak.	—	−141.72	—	−152.18
Montgomery, Ala.	7.35	128.70	−10.93	65.64
San Bernardino, Calif.	—	−115.69	—	−106.01
Miami, Fla.	−94.32	−114.94	−105.51	−87.56
Toledo, Ohio	22.09	103.06	11.03	97.89
Albany, N.Y.	—	94.79	—	76.00
Ten largest 1969 residuals (age-sex-race-adjusted total mortality rates)				
Duluth, Minn.	62.35	209.72	48.07	161.62
Fargo, N.Dak.	—	−141.72	—	−152.18
Honolulu, Hawaii	—	−181.21	—	−135.48
Wilkes-Barre, Pa.	225.74	175.38	195.06	−122.90
San Bernardino, Calif.	—	−115.69	—	−106.01
Toledo, Ohio	22.09	103.06	11.03	97.89
Erie, Pa.	—	93.62	—	97.25
Johnstown, Pa.	27.99	91.35	33.99	95.97
Mobile, Ala.	19.05	−81.51	26.73	−94.05
San Jose, Calif.	−77.74	−53.92	−101.29	−89.74

Note: The second column shows residuals from the 1960 unadjusted total mortality rate equation (regression 7.1-1) based on 117 SMSAs, the third column shows residuals from the corresponding 1969 replication (regression 7.1-3) based on 112 SMSAs, the fourth column shows residuals from the 1960 adjusted total mortality rate equation (regression 7.1-6) based on 117 SMSAs, and the fifth column shows residuals from the corresponding 1969 replication (regression 7.1-8) based on 112 SMSAs. The standard deviations for the four sets of residuals were as follows: 63.89 (regression 7.1-1), 65.99 (regression 7.1-3), 63.15 (regression 7.1-6), and 54.43 (regression 7.1-8).

A negative residual indicates that the regression equation overestimated the mortality rate for the particular SMSA.

From: Lave and Seskin (1977), page 133.

positive associations between suspended sulfates and mortality in 1960, 1969, and 1974 and (2) significant positive associations for TSP in 1960 and 1969, but not 1974 against the following results reported by others:

(1) Results reported by Mendelsohn and Orcutt (1979), based on regression analyses of associations between 1970 mortality rates (for 404 country groups throughout the United States) and air pollution exposures retrospectively estimated on the basis of 1970 and 1974 annual average pollutant data from air monitoring sites in the same or nearby counties, which suggested fairly consistent (though variable) associations between mortality for some age groups (increasingly more positive with age) and sulfate levels but much less consistent and sometimes negative associations with TSP or other pollutant levels.

(2) Results obtained by Thibodeau et al. (1980), interpreted as not disagreeing with the conclusions of Lave and Seskin concerning mortality being associated with chronic exposure to PM, SO_2 or sulfates, but also accompanied by indications that the regression coefficients for air pollution variables are quite unstable and should be used with great care when interpreting their meaning.

(3) Results reported by Lipfert (1980), derived from an analysis taking into account a smoking index based on state tax receipts, which Lipfert interpreted as showing sulfates to be least harmful of seven air pollutants (including SO_2 and TSP), although no adjustments for urban-rural differences in study population residences were used.

(4) Analyses of 1970 United States mortality data by Crocker et al. (1979), taking into account retrospectively estimated nutritional variables and a smoking index, but indicating no significant relationships between air pollution and total mortality.

(5) Results of Gerking and Schultz (1981), using the same data base, that indicated a significant positive relationship between TSP and total mortality when using an OLS model similar to that of Lave and Seskin (1977) but finding negative, though not significant, air pollution coefficients after adding smoking, nutrition, exposure-to-cold, and medical-care variables to a two-equation model.

Various criticisms of each of the above studies could be presented, as cited in one or more of the other respective studies or Chappie and Lave (1981), but to little avail at this time in trying to ascertain which findings may be "more valid" than others. Thus, although many of the studies qualitatively suggest some positive associations between mortality and chronic exposure to certain air pollutants (e.g., sulfates in the case of several studies) in the United States, many issues remain unresolved concerning any such associations. These include questions regarding the strength and stability of the reported air pollution-mortality relationships across geographic areas, time periods, ranges of pollutant exposure levels, and variations in other variables or combinations of variables included in the underlying analyses. Furthermore, to the extent that the analyses generally do not adqeuately rule out the possibility of reported associations being due to potentially important unaccounted-for variables that may covary with the air pollution gradients, then it is not possible to discern whether the reported associations are either merely fortuitous on the one hand or, perhaps, causal on the other hand.

Other epidemiological studies have more specifically attempted to relate lung cancer mortality to chronic exposures to sulfur oxides, PM undifferentiated by chemical composition, or specific PM chemical species. However, little or no clear epidemiological evidence has been advanced to date to substantiate hypothesized links between SO_2 or other sulfur oxides and cancer; nor does there presently exist credible epidemiological evidence linking increased cancer rates to elevations in PM as a class, i.e., undifferentiated as to chemical content. See reviews by Higgins (1976) and Doll (1978) evaluating associations between cancer mortality and environmental exposures to various air pollutants. Other epidemiological studies (e.g., of occupationally-exposed workers) also provide some evidence of increased cancer risk associated with exposure to certain types of PM, e.g., certain organic compounds or metals, often found in the fine- and coarse-mode particulate fractions of many urban aerosols (see Appendix 14B). However, no well-accepted basis currently exists for quantitatively defining any consistent relationships concerning relative contributions or levels of such PM components to possible carcinogenic effects of PM pollution as a whole.

14.4.2 Morbidity

14.4.2.1 Respiratory Effects in Adults--Impairment of pulmonary function is likely to be one of the effects of exposures to air pollution, since the pulmonary system includes tissues that receive the initial impact when toxic materials are inhaled. Acute and chronic changes in function may be significant biological responses to air pollution exposure. A number of studies have been conducted in an effort to relate pulmonary function changes to the presence of air pollutants in European, Japanese, and American communities. However, few provide more than qualitative evidence relating pulmonary function changes to elevations in SO_2 or PM (see Appendix 14A).

A series of studies, reported on from the early 1960s to the mid-1970s were conducted by Ferris, Anderson, and others (Ferris and Andersen, 1962; Kenline, 1962; Andersen et al., 1964; Ferris et al., 1967, 1971, 1976). The initial study involved comparison of three areas within a pulp-mill town (Berlin, New Hampshire). Kenline (1962) reported average 24-hour SO_2 levels during a limited sampling period in the summer (August-September 1960) to be only 16 ppb and average 24-hour TSP levels for the two-month period to be 183 $\mu g/m^3$. In the original prevalence study (Ferris and Anderson, 1962; Anderson et al., 1964), no association was found between questionnaire-determined symptoms and lung function tests assessed in the winter and spring of 1961 in the three areas with differing pollution levels after standardizing for cigarette smoking. The authors discuss why residence is a limited indicator for exposure (Anderson et al., 1964). The study was later extended to compare Berlin, New Hampshire, with the cleaner city of Chilliwack, British Columbia in Canada (Anderson and Ferris, 1965). Sulfation rates (lead candle method) and dustfall rates were higher in Berlin than in Chilliwack. The prevalence of chronic respiratory disease was greater in Berlin, but the authors concluded that this difference was due to the interaction between age and smoking habits within the respective populations. Higher levels of respiratory function in some cigarette-smoking groups in the cleaner area were observed, but this difference could be due to socioeconomic and ethnic differences as well as air pollution (Higgins, 1974).

The Berlin, New Hampshire, population was followed up in 1967 and again in 1973 (Ferris et al., 1971, 1976). During the period between 1961 and 1967, all measured indicators of air pollution fell, e.g. TSP from approximately 180 μg/m^3 in 1961 to 131 μg/m^3 in 1967. In the 1973 follow-up, sulfation rates nearly doubled from the 1967 level (0.469 to 0.901 mg SO_3/100 cm^2/day) while TSP values fell from 131 to 80 μg/m^3. According to WHO (1979) only limited data on SO_2 was available (the mean of a series of 8-hour samples for selected weeks). During the 1961 to 1967 period, standardized respiratory symptom rates decreased and there was an indication that lung function also improved. Between the period 1967 to 1973, age-sex standardized respiratory symptom rates and age-sex-height standardized pulmonary function levels were unchanged. Although some of the testing was done during the spring versus the summer in the different comparison years, Ferris and coworkers attempted to rule out likely seasonal effects by retesting some subjects in both seasons during one year and found no significant differences in test results. Given that the same set of investigators, using the same standardized procedures, conducted the symptom surveys and pulmonary function tests over the entire course of these studies, it is unlikely that the observed health endpoint improvements in the Berlin study population were due to variations in measurement procedures, but rather appear to be associated with decreases in TSP levels from 180 to 131 μg/m^3. The relatively small changes observed and limited aerometric data available, however, argue for caution in placing much weight on these findings as quantitative indicators of "effect" or "no-effect" levels for health changes in adults associated with chronic exposures to PM measured as TSP.

One other American study was found to provide potentially useful qualitative or quantitative information regarding association of morbidity effects in adults with ambient exposures to SO_2 or particulate matter. A cross-sectional study was conducted by Bouhuys (1978) in two Connecticut communities in which differences in respiratory and pulmonary function were examined in 3056 subjects (adults and children). Hosein (1977a) reported on aerometric data used in the study, which were obtained at three sites in Ansonia (urban) and four sites in Lebanon (rural) near the residences of study subjects. The TSP levels during the period of the study in Lebanon and Ansonia were 39.5 and 63.1 μg/m^3 and SO_2 levels were 10.9 and 13.5 μg/m^3, respectively. Site-to-site variations on the same day were frequently significant in Ansonia and also occurred in Lebanon. During the years 1966-72, annual average TSP levels in Ansonia ranged from 88 to 152 μg/m^3. No historical data for SO_2 or TSP in Lebanon were provided. Size fractionation (Hosein, 1977b) of a limited number of TSP samples in Ansonia showed that 81 percent of the TSP sample was 9.4 μm or less in diameter. Binder et al. (1976) obtained for 20 subjects in Ansonia one 24-hour measure of personal air pollution exposure for: particles ($\leq$ 7 μm diameter), SO_2, and NO_2. Subjects with smokers in the home were exposed to significantly higher levels than those without such exposure. Personal exposure and outdoor exposures were also significantly different. The mean personal respiratory level was 114 $\mu g/m^3$ as compared to the outdoor TSP level of 58.4 $\mu g/m^3$.

An extended version of the MRC Questionnaire was administered via a computer data-acquisition terminal (Mitchell, 1976) between October 1972 and January 1973 in Lebanon and from mid-April through July 1973 in Ansonia. For children (7-14 yrs) the response rate varied from 91-96 percent for boys and girls. For adults (25-64 years) the response rate was 56 percent in Ansonia and 80 percent in Lebanon. After analysis of non-responder versus responder differences, the responders were considered to be representative of the total population, although some significant differences were noted between responders and non-responders for some symptom reporting and current smoking in some age groups.

Bouhuys (1978) found no differences between Ansonia and Lebanon for chronic bronchitis prevalence rates but did note that a history of bronchial asthma was highly significant for male residents of Lebanon (the cleaner town) as compared to Ansonia (the higher pollution area). No differences were observed between the communities for pulmonary function tests adjusted for sex, age, height and smoking habits. However, three out of five symptoms (cough, phlegm, and plus one dyspnea) prevalences were significantly higher for adult non-smokers in Ansonia ($P < .001$). The inconsistencies apparent in terms of both positive and negative health effect results obtained in this cross-sectional study make it difficult to interpret. Although it appears to be a generally negative study, the significantly increased symptom rates raise questions as to whether some impact on health might have occurred. A follow-up longitudinal examination could have determined whether the effects persisted. Also, it is not clear whether the reported effects relate to current or historical pollutant levels, and occupational exposure was not examined. The seasonal difference in conduct of the two surveys might have also had some effect on the acquired health data. A later study (Bouhuys et al. 1979) compared results obtained in the Connecticut communities with results obtained by analogous testing procedures in Winnsboro, South Carolina, and found no signigicant differences between results obtained in Ansonia versus those obtained in Winnsboro where air pollution was markedly lower. However, because of differences in racial mixes between the two communities and other demographic differences not properly controlled for, these results cannot be accepted as adequately demonstrating no differences in health effects between the towns.

14.4.2.2 Respiratory Effects in Children--Numerous epidemiological studies have attempted to evaluate possible relationships between chronic exposure to air pollution containing PM and SO_2 and the occurence of health-related changes in children. However, very few studies provide adequate evidence qualitatively establishing such relationships; and still fewer provide dose-response (or exposure-response) data of a type or of sufficient quality by which to estimate even approximate chronic PM or SO_2 exposure levels associated with changes in health status in children.

An apparent quantitative relationship between air pollution and lower respiratory tract illness in children was demonstrated by Lunn et al. (1967). These investigators studied respiratory illness in 5- and 6-year-old schoolchildren living in four areas of Sheffield, England. Air pollution concentrations showed a gradient in 1964 across four study areas for mean 24-hour smoke (BS) concentrations from 97 $\mu g/m^3$ to 301 $\mu g/m^3$ and the same gradient for annual mean 24-hour SO_2 concentrations from 123 $\mu g/m^3$ to 275 $\mu g/m^3$. The following year, the annual concentrations of smoke were about 20 percent lower and SO_2 about 10 percent higher, but the gradient was preserved for each pollutant. In high-pollution areas, individual 24-hour mean smoke concentrations exceeded 500 $\mu g/m^3$ 30 to 45 times in 1964 and 0 to 15 times in 1965 for the lowest and highest pollution areas, respectively. Sulfur dioxide exceeded 500 $\mu g/m^3$ 11 to 32 times in 1964 and 0 to 23 times in 1965 for the lowest and highest pollution areas, respectively. Information on respiratory symptoms and illness was obtained by questionnaires completed by the parents, by physical examination, and by tests of pulmonary function ($FEV_{0.75}$ and FVC). Socioeconomic factors (SES) were considered in the analyses, but home-heating systems were not. Although certain differences in SES between areas were noted, the gradients between areas existed even when the groups were divided by social class, number of children in house, and so on. Positive associations were found between air pollution concentrations and both upper and lower respiratory illness. Lower respiratory illness was 33 to 56 percent more frequent in the higher pollution areas than in the low-pollution area ($p < 0.005$). Also, decrements in lung function as measured by spirometry tests were closely associated with the occurrence of respiratory disease symptoms. The main respiratory symptom results observed by Lunn et al. (1967) are illustrated in Figure 14-4.

The authors of the study (Lunn et al., 1967) highlighted the following points in discussing their results:

> "The respiratory measurement findings showed no association with area, social class, children in the house, and sharing of bedrooms, although Attercliffe, the area of highest pollution, showed reduced $F.E.V._{0.75}$ and F.V.C. ratios. On the other hand, very clear evidence of reduced $F.E.V._{0.75}$ ratios emerged where there was a past history of pneumonia and bronchitis, persistent or frequent cough, or colds going to the chest. It must be stressed that these findings relate to first year infant schoolchildren and that measurements were made during the summer term when pollution levels were low and acute respiratory infections few and far between. In other words, a pattern of respiratory disability had appeared at an early age and was sufficiently established to persist although the factors of pollution and infection were temporarily absent or at a low level."

In a second report, Lunn et al. (1970) gave results for 11-year-old children studied in 1963-64 that were similar to those provided earlier for the younger group. Upper and lower respiratory illness occurred more frequently in children exposed to annual average 24-hour mean smoke (BS) concentrations of 230 to 301 $\mu g/m^3$ and 24-hour mean SO_2 concentrations of 181-275 $\mu g/m^3$ than in children exposed to smoke (BS) at 97 $\mu g/m^3$ and SO_2 at 123 $\mu g/m^3$. This report also provided additional information obtained in 1968 on 68 percent of the children who

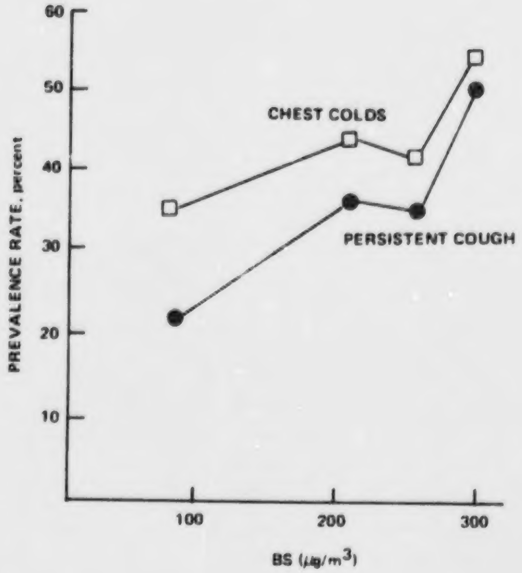

Figure 14-4. History and clinical evidence of respiratory disease (percent) in 5-year-olds, by pollution in area of residence. BS (μg/m^3) levels indicated above must be taken as only very crude approximations of actual PM mass present due to ambiguities regarding use of site-specific calibrations in deriving the mass estimates.

Source: Lunn et al. (1967).

were 5 and 6 years old in 1963-64. By 1968, the reported concentrations of smoke (BS)* were only about one-half those measured in 1964, SO_2 concentrations were about 10 to 15 percent below those measured in 1964, and the pollution gradient no longer existed, so the combined three higher pollution areas were compared with the single original low-pollution area. Lower respiratory illness prevalence measured as "colds going to chest" was 27.9 percent in the low-pollution area and 33.3 percent in the combined high-pollution areas, but the difference was not statistically significant ($p > 0.05$). Ventilatory function results were similar. Also, the 9-year-old children had less respiratory illness than the 11-year-old group seen previously. Since 11-year-old children generally have less respiratory illness than do 9-year-olds, this represented an anomaly that the authors suggested may have been the result of improved air quality.

It should be noted that these Lunn et al. (1967, 1970) findings have been widely accepted (Rall, 1974; Higgins, 1974; Holland et al., 1979; National Research Council, 1978a,b; Ferris, 1978; WHO, 1979) as being valid; and, on the basis of the results reported, it appears that increased frequency of lower respiratory symptoms and decreased lung function in children may occur with long-term exposures to annual BS levels in the range of 230-301 $\mu g/m^3$ and SO_2 levels of 181-275 $\mu g/m^3$. However, these must be taken only as very approximate "observed effect" levels due to uncertainties associated with estimating PM mass based on BS readings. Also, one cannot conclude based on the 1968 follow-up study results, that "no-effect" levels were demonstrated for BS levels in the range of 48-169 $\mu g/m^3$ and SO_2 levels in the range of 94-253 $\mu g/m^3$ due to: (1) the likely insufficient power of the study to have detected small changes given the size of the population cohorts studied; and (2) the lack of site-specific calibration of the BS mass readings at the time of the later (1968) study. The lack of observed effects in this follow-up study suggests, however, that the retested children may have recovered from air pollution-induced decrements in lung function detected by the earlier study (Lunn et al., 1967).

In summary, the one above study by Lunn et al. (1967) in Sheffield, England, provides the clearest evidence for association of significant pulmonary function decrements and increased respiratory disease illnesses in children with chronic exposure to specific levels of SO_2 and PM in the ambient air.

14.5 SUMMARY AND CONCLUSIONS

Some epidemiological studies reviewed above appear to provide meaningful quantitative information on health effects associated with ambient air exposures to PM and SO_2. Others, however, either do not fully meet the various objectives discussed earlier under Section 14.1.2 or ambiguity exists regarding clear interpretation of their reported results. Only some study results can, therefore, be accepted with a relatively high degree of confidence, whereas others may provide, at best, only suggestive evidence for the reported associations between air pollution and health effects. The main focus of this section is on summarizing and interpreting results from those selected key studies yielding results that can now be accepted with the greatest degree of certainty. However, even those findings must be viewed as providing only very approximate estimates of ambient air levels of PM and SO_2 likely to be associated with the health effects indicated; and the individual contributions of PM and SO_2 cannot be clearly separated from the effects of each other or other covarying air pollutants.

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*However, in contrast to the reported smoke concentrations (in $\mu g/m^3$) for 1963 being based on site-specific calibrations carried out in Sheffield in 1963, the later smoke estimates reported for 1968 were not derived from any subsequent recalibration of the PM mass to BS reflectance relationship. Such recalibration would be necessary to adjust for changes in atmospheric PM chemical composition (especially the proportion of elemental carbon) likely to have resulted from decreased emissions of partially combusted fossil fuel products from open-hearth burning of coal, which markedly decreased in Sheffield from 1963 to 1968.

In general, the epidemiological studies reviewed here provide strong evidence for induction by marked elevations of atmospheric levels of PM and SO_2 of severe health effects, such as mortality and respiratory diesease, in certain populations at special risk. Populations at special risk for such effects appear to include, mainly, the elderly and adults with chronic preexisting cardiac or respiratory diseases (e.g., bronchitics). Increased respiratory tract illnesses and more transient effects, e.g., decrements in pulmonary function, also appear to be associated for children with lower chronic exposures to SO_2 and PM.

14.5.1 Health Effects Associated with Acute Exposures to Particulate Matter and Sulfur Oxides

As noted earlier in the present chapter, it is widely accepted that increases in mortality occur when either SO_2 or PM (as BS) levels increase beyond 24-hour levels of 1000 $\mu g/m^3$. Such increased mortality, mainly in the elderly or chronically ill, might logically be attributed in part to even brief exposures to very high short-term peak (hourly) levels in the pollutants, which at times increased to several thousand $\mu g/m^3$ during certain major pollution episodes. However, none of the available epidemiological data have been collected or analyzed in a manner so as to either credibly substantiate or refute this possibility. Much more clearly established are marked increases in mortality and morbidity being associated with prolonged episodic elevations of PM and SO_2 which average out to daily levels of 1000 $\mu g/m^3$, especially in the presence of high humidity (fog) conditions, but which include continuous exposures to high pollutant (PM and SO_2) concentrations for several days without intermittent relief or return to near normal levels at points between short-term pollution peaks. Thus, although 24-hour concentrations of PM and SO_2 $\geq$1000 $\mu g/m^3$ can be stated as levels at which mortality has notably increased, great care must be exercised in generalizing these observations in attempting to predict likely effects associated with comparable elevations at other times and locations. In particular, the prolonged or continuous nature of the high pollutant exposures and other interacting factors present, e.g., high humidity levels, must be taken into account as additional important determinants of mortality increases observed so far during major air pollution episodes; and marked increases in mortality should not be expected to occur regularly as a function of short-term peak excursions of 24-hour PM or SO_2 levels above 1000 $\mu g/m^3$. Consistent with this are numerous examples in the epidemiological literature evaluated above where no detectable increases in mortality were found to occur on various scattered days when PM and/or SO_2 levels reached comparably high ($\geq$1000 $\mu g/m^3$) 24-hour levels as on other days (or sets of successive days) when mortality was clearly increased.

Even more difficult to establish are to what extent smaller but significant increases in mortality and morbidity are associated with nonepisodic 24-hour average exposures to SO_2 and/or levels below 1000 $\mu g/m^3$. Concisely summarized in Table 14-7 are findings from several key studies reviewed above which appear to demonstrate with a reasonably high degree of certainty mortality and morbidity effects associated with acute exposures (24 hrs) to these pollutants. The first two studies cited, by Martin and Bradley (1960) and Martin (1964), deal with a relatively small body of data from London in the late 1950s. No clear "threshold" levels were revealed by their analyses regarding SO_2 or BS levels at which significantly increased mortality began to occur. However, based on their findings, and reanalyses of the Martin and Bradley data by Ware et al. (1981), mortality in the elderly and chronically ill was clearly elevated in association with exposure to ambient air containing simultaneous SO_2 and BS levels above 1000 $\mu g/m^3$; and some indications exist from these analyses that slight increases in mortality may have been associated with nonepisodic BS and PM levels in the range of 500-1000 $\mu g/m^3$ (with greatest certainty demonstrated for levels in excess of 750 $\mu g/m^3$). Much less certainty is attached to suggestions of mortality increases at lower levels, possibly based on the Ware et al. (1981) or other reanalyses (Appendix 14E) of the Martin and Bradley data, especially in view of wide 95 percent confidence intervals demonstrated by one analysis (Appendix 14D) to be associated with estimation of dose-response relationships between BS or SO_2 and mortality using the Martin and Bradley (1960) data. Analyses by Mazumdar et al. (1981) for 1958-59 to 1971-72 are generally consistent with the above findings but seem to suggest that the 1958-59 London winter may represent something of a worst-case situation in comparison to most later winters. Still the Mazumdar et al. (1981) and certain other analyses (Appendix 14F) of 1958-59 to 1971-72 London winter mortality data are strongly indicative of small, but significant increases in mortality occurring at BS levels below 500 $\mu g/m^3$ and, possibly, as low as 150 to 200 $\mu g/m^3$.

Only very limited data exist by which to attempt to delineate any specific physical and chemical properties of PM associated with the observed increases in mortality. Based on information noted earlier (Section 14.2), it would seem that marked increases in small particles to levels above 500-1000 $\mu g/m^3$ appear to be most clearly associated with increased mortality, based on the BS aerometric measurements reported, although contributions from larger coarse-mode particles cannot be completely ruled out. Nor is it possible to state with certainty specific PM chemical species associated with the increases in mortality. We do know that large amounts of pollutants (e.g., elemental carbon, tarry organic matter, etc.) from incomplete combustion of coal were present in the air and mortality levels appeared to decrease as PM concentrations declined over the years; but no single component or combinations of particulate pollutants can clearly be implicated. Neither can the relative contributions

TABLE 14-7. SUMMARY OF QUANTITATIVE CONCLUSIONS FROM EPIDEMIOLOGICAL STUDIES RELATING HEALTH EFFECTS TO ACUTE EXPOSURE TO AMBIENT AIR LEVELS OF SO_2 AND PM

Type of study	Effects observed	24-hr average pollutant level ($\mu g/m^3$)		Reference
		BS	SO_2	
Mortality	Clear increases in daily total mortality or excess mortality above a 15-day moving average among the elderly and persons with preexisting respiratory or cardiac disease during the London winter of 1958-59.	$\geq$1000	$\geq$1000	Martin and Bradley (1960); Martin (1964)
	Analogous increases in daily mortality in London during 1958-59 to 1971-72 winters.			Mazumdar et al. (1981)
	Some indications of likely increases in daily total mortality during the 1958-59 London winter, with greatest certainty (95% confidence) of increases occurring at BS and SO_2 levels above 750 $\mu g/m^3$.	500-1000	500-1000	Martin and Bradley (1960)
	Analogous indications of increased mortality during 1958-59 to 1971-72 London winters, again with greatest certainty at BS and SO_2 levels above 750 $\mu g/m^3$ but indications of small increases at BS levels <500 $\mu g/m^3$ and possibly as low as 150-200 $\mu g/m^3$.			Mazumdar et al. (1981)
Morbidity	Worsening of health status among a group of chronic bronchitis patients in London during winters from 1955 to 1960.	$\geq$250-500*	$\geq$500-600	Lawther (1958); Lawther et al. (1970)
	No detectable effects in most bronchitics; but positive associations between worsening of health status among a selected group of highly sensitive chronic bronchitis patients and London BS and SO_2 levels during 1967-68 winter.	<250*	<500	Lawther et al. (1970)

*Note that the 250-500 $\mu g/m^3$ BS levels stated here may represent somewhat higher PM concentrations than those actually associated with the observed effects reported by Lawther (1970). This is due to the estimates of PM mass (in $\mu g/m^3$ BS) used by Lawther being based on the D.S.I.R. calibration curve found by Waller (1964) to approximate closely a site-specific calibration curve developed by Waller in central London in 1956, but yielding somewhat higher mass estimates than another site-specific calibration developed by Waller a short distance away in 1963. However, the precise relationship between estimated BS mass values based on the D.S.I.R. curve versus the 1963 Waller curve cannot be clearly determined due to several factors, including the non-linearity of the two curves and their convergence at low BS reflectance values.

of SO_2 or PM be clearly separated based on these study results nor can possible interactive effects with increases in humidity (fog) be completely ruled out. Temperature change, however, does not appear to be a key determinant in explaining mortality effects demonstrated by the above analyses to be associated with atmospheric elevations of PM or SO_2.

Studies conducted in New York City by Greenburg and coworkers (and other groups of investigators) appear to suggest, with less confidence, that slight mortality increases may have been associated with episodic increases in 24-hour SO_2 levels above 1000 $\mu g/m^3$ and CoHs levels of 5.0-7.0, but little credible evidence exists for mortality having occurred with lower acute (24-hour) exposures. Again, specific particulate chemical species cannot be clearly implicated nor the relative contributions of SO_2 and particulate matter separated in regard to any induction of mortality by episodic pollution. It should be noted that, whatever the causal agents, only very small increases in mortality may have been detected at the above pollutant levels in New York City; and associated morbidity effects appear to have been restricted mainly to increased respiratory or cardiac complaints among the elderly.

Similar analysis of the Lawther morbidity studies listed in Table 14-7 suggests that acute exposure to elevated 24-hour PM levels in the range of 250-500 $\mu g/m^3$ (BS) in association with 24-hour SO_2 levels of 500-600 $\mu g/m^3$ were most clearly associated with the induction of respiratory disease symptoms among large (>1000) populations of chronically ill London bronchitis patients. A smaller population (~80) of selected, highly sensitive London bronchitic patients appeared to be affected at somewhat lower BS and SO_2 levels, but specific exposure-effect levels could not be determined on the basis of the reported data. Again, however, little can be said in terms of specifying physical or chemical properties of PM associated with these observed morbidity effects beyond the comments noted above in relation to Martin's studies on mortality.

14.5.2 Health Effects Associated with Chronic Exposures to PM and SO_x

In regard to chronic exposure effects of SO_2 and particulate matter, the best pertinent epidemiological health studies are summarized in Table 14-8. The studies by Ferris et al. (1973, 1976) suggest that lung function decrements may occur in adults at TSP levels in excess of 180 $\mu g/m^3$ in the presence of relatively low estimated SO_2 levels, whereas no effects were observed by the same investigators at TSP levels below 130 $\mu g/m^3$. Other studies listed in Table 14-8 suggest that significant respiratory effects occur in children in association with long-term (annual average) PM levels in the approximate range of 230-301 $\mu g/m^3$ (BS) in association with SO_2 levels of 181-275 $\mu g/m^3$.

No specific particulate matter chemical species can clearly be implicated as causal agents associated with the effects observed in the studies listed in Table 14-8. Nor can potential contributions of relatively large inhalable coarse mode particles be ruled out based on these study results. It should be remembered that various occupational studies listed in Appendix 14B at least qualitatively suggest that such sized particles of many different types of chemical composition can be associated with significant pulmonary decrements, respiratory tract pathology, and morphological damage.

TABLE 14-8. SUMMARY OF QUANTITATIVE CONCLUSIONS FROM EPIDEMIOLOGICAL STUDIES RELATING HEALTH EFFECTS TO CHRONIC EXPOSURE TO AMBIENT AIR LEVELS OF SO_2 AND PM

Type of study	Effects observed	Annual average pollutant levels ($\mu g/m^3$) particulate matter BS	 TSP	 SO_2	Reference
Cross-sectional (4 areas)	Likely increased frequency of lower respiratory symptoms and decreased lung function in children in Sheffield, England	230-301*	-	181-275	Lunn et al. (1967)
Longitudinal and cross-sectional	Apparent improvement in lung function of adults in association with decreased PM pollution in Berlin, N.H.	-	180	**	Ferris et al. (1973, 1976)
Longitudinal and cross-sectional	Apparent lack of effects and symptoms, and no apparent decrease in lung function in adults in Berlin, N.H.		80-131	**	Ferris et al. (1973, 1976)

*Note that BS levels stated here in $\mu g/m^3$ must be viewed as only crude estimates of the approximate PM (BS) mass levels associated with the observed health effects, given ambiguities regarding the use or non-use of site-specific calibrations in Sheffield to derive the reported BS levels in $\mu g/m^3$.

**Note that sulfation rate methods indicated low atmospheric sulfur levels in Berlin, N.H. during the time of these studies. Crude estimation of SO_2 levels from that data suggest that <25-50 $\mu g/m^3$ SO_2 levels were generally present in Berlin, N.H., and did not likely contribute to observed health effects.

Only very limited information has been published (Commins and Waller, 1967) on the chemical composition of particulate matter present in London air during the period of some of the above epidemiological studies demonstrating associations between mortality or morbidity effects and elevations in PM levels, as summarized in Table 14-9. Such data may provide important clues as to possible causative agents involved in the etiology of health effects observed in London during the 1950s and early 1960s. For the sake of comparison, information on measured chemical components of TSP matter in U.S. cities during the early 1960s is also provided in Table 14-9. It must be noted, however, that likely substantial differences in specific components of the PM present in London air of the 1950's and 1960's versus the chemical composition of PM currently present in urban aerosols over American cities argue for much caution in extrapolating results of London epidemiological studies for present criteria development purposes.

TABLE 14-9. COMPARISON OF MEASURED COMPONENTS OF TSP IN U.S. CITIES (1960-1965) AND MAXIMUM 1-HOUR VALUES IN LONDON (1955-1963)

	UNITED STATES[a]			LONDON[b]
		Concentration µg/m^3		
Pollutant	Number of stations	Arith. average	Maximum 24-hour	Maximum 1-hour[c]
Suspended Particles	291	105	1254 (TSP)	9700 (Smoke)
Fractions:				
Benzene-soluble organics	218	6.8		
Chloride (water soluble)	-	-	-	410
Nitrates	96	2.6	39.7	5
Sulfates	96	10.6	101.2	666
Sulfuric acid	-	-	-	680
Ammonium	56	1.3	75.5	
Antimony	35	0.001	0.160	<1
Arsenic	133	0.02		
Beryllium	100	<0.0005	0.010	<1
Bismuth	35	<0.0005	0.064	<1
Cadmium	35	0.002	0.420	1
Calcium	-	-	-	32
Chromium	103	0.015	0.330	2
Cobalt	35	<0.0005	0.060	<1
Copper	103	0.09	10.00	2
Iron	104	1.58	22.00	25
Lead	104	0.79	8.60	22
Manganese	103	0.10	9.98	5
Molybdenum	35	<0.005	0.78	<1
Nickel	103	0.034	0.460	1
Tin	85	0.02	0.50	2
Titanium	104	0.04	1.10	1
Vanadium	99	0.050	2.200	2
Zinc	99	0.67	58.00	24
Gross beta radioactivity	323	(0.8 pCi/m^3)	(12.4 pCi/m^3)	

[a] U.S. Dept. Health, Education, & Welfare, 1970.

[b] Commins and Waller, 1967.

[c] Obtained from one London site.

14.6 REFERENCES*

American Thoracic Society. Epidemiology Standardization Project. Am. Rev. Res. Dis. 118(6,pt.2), 1978.

Anderson, D. O., and A. A. Larsen. The incidence of illness among young children in two communities of different air quality: A pilot study. Can. Med. Assoc. J. 95:893, 1966.

Anderson, D. O., and B. G. Ferris, Jr. Air pollution levels and chronic respiratory disease. Arch. Environ. Health 10:307-311, 1965.

Anderson, D. O., B. G. Ferris, Jr., and R. Zinkmantel. Levels of air pollution and respiratory disease in Berlin, New Hampshire. Am. Rev. Respir. Dis. 90:877-887, 1964.

Angel, J. H., C. M. Fletcher, I. D. Hill, and C. M. Finker. Respiratory illness in factory and office workers. Br. J. Dis. Chest 59: 66-80, 1965.

Ashley, D. J. B. The distribution of lung cancer and bronchitis in England and Wales. Brit. J. Cancer 21: 243, 1967.

Aubrey, F., G. W. Gibbs, and M. R. Becklake. Air Pollution and Health in three urban communities. Arch. Environ. Health 34:360-368, Sept/Oct 1979.

Ball, D. J., and R. Hume. The relative importance of vehicular and domestic emissions of dark smoke in Greater London in the mid-1970's, the significance of smoke shade measurements, and an explanation of the relationship of smoke shade to gravimetric measurements of particulate. Atmos. Environ. 11:1065-1073, 1977.

Bates, D. V. Air pollution and chronic bronchitis. Arch. Environ. Health 14:220, 1967.

Bates, D. V. The fate of the chronic bronchitic: A report of the ten-year followup in the Canadian Department of Veterans Affairs Coordinated Study of Chronic Bronchitis. Am. Rev. Res. Dis. 108:1043, 1973.

Bates, D. V., C. A. Gordon, G. I. Paul, R. E. G. Place, D. P. Snidal, and C. R. Woolf. (with special sections contributed by M. Katz, R. G. Fraser, and B. B. Hale) Chronic bronchitis. Report on the third and fourth stages of the Coordinated Study of Chronic Bronchitis in the Department of Veterans Affairs. Canada. Med. Serv. J. Can. 22:5, 1966.

Bates, D. V., C. R. Woolf, and G. I. Paul. Chronic bronchitis: A report on the first two stages of the Coordinated Study of Chronic Bronchitis in the Department of Veterans Affairs. Canada. Med. Serv. J. Can. 18:211, 1962.

Becklake, M. R., J. Soucie, G. W. Gibbs, and H. Ghezzo. Respiratory health status of children in three Quebec urban communities. An Epidemiologic Study. Bull. Europ. Physiopath. Resp. 14:205-221, 1978.

Becker, W. H., F. J. Schilling, M. P. Verma. The effect on health of the 1966 Eastern Seaboard air pollution episode. Arch. Environ. Health 16:414-419, 1968.

Bennett, A. E., W. W. Holland, T. Halil, and A. Elliot. Lung function and air pollution. Chronic inflammation of the bronchi. Prog. Respir. Res. 6:78-89, 1971.

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*Note that references are included here for studies discussed in accompanying Chapter 14 appendix materials, as well as for studies cited in the main text of Chapter.

Biersteker, K. Polluted Air Causes, Epidemiological Significance, and Prevention of Atmospheric Pollution. Assen, Netherlands, Van Gorcum and Co., pp. 21-23 (in Dutch), 1966.

Biersteker, K., and P. van Leeuwen. Air pollution and peak flow rates of schoolchildren in two districts of Rotterdam. Arch. Environ. Health 20:382-384, 1970.

Biersteker, K., and P. van Leeuwen. Air pollution, bronchitis prevalence and peak flow rates of schoolchildren in two districts of Rotterdam (Netherlands). In: 2nd Int. Clean Air Cong. Proc. Washington, D.C., December 1970. H. M. Englund and W. T. Berry (ed.). New York, Academic Press. p. 209-212.

Binder, R. E., C. A. Mitchell, H. R. Hosein, and A. Bouhuys. Importance of the Indoor Environment in Air Pollution Exposure, Arch. of Environ. Health. Nov./Dec. 277-279, 1976.

Bouhuys, A., G. J. Beck, and J. B. Schoenberg. Do present levels of air pollution outdoors affect respiratory health? Nature 276:466-471, 1978.

Bouhuys, A., G. J. Beck, and J. B. Schoenberg. Priorities in prevention of chronic lung diseases. Lung, 156:129-148, 1979.

British Standards Institution. Methods for the measurements of air pollution, Part 2. Determination of Concentration of Suspended Matter. British Standard 1747: Part 2, 1964.

Buck, S. F., and A. D. Brown. Mortality from Lung Cancer and Bronchitis in Relation to Smoke and Sulfur Dioxide Concentration, Population Density, and Social Index. Research Paper No. 7. London, Tobacco Research Council. 1964.

Buechley, R. W. SO_2 levels, 1967-1972 and perturbation in mortality. A further study in the New York-New Jersey Metropolis. Presented to the Health Research Council of the City of New York Meeting, The Environmental Pollution Impact of Stationary Source Fossil Fuel Air Pollution Control Measures in New York City, March 6, 1975.

Buechley, R. W. SO_2 Levels, 1967-1972 and Perturbations in Mortality. Contract No. ES-5-2101. Report available from National Institute of Environmental Health Sciences, Research Triangle Park, N.C., 1977.

Buechley, R. W., W. B. Riggan, W. Hasselblad, and J. B. Van Bruggen. SO_2 levels and perturbations in mortality. A study in New York-New Jersey metropolis. Arch. Environ. Health 27:134-137, 1973.

Burgess, S. E., and C. W. Shaddick. Bronchitis and air pollution. R. Soc. Health J. 79:10-24, 1959.

Burn, J. L., and J. Pemberton. Air pollution bronchitis and lung cancer in Salford. Int. J. Air Water Pollut. 7:5-16, 1963.

Burrows, B., A. L. Kellogg, and J. Bushey. Relationship of symptoms of chronic bronchitis and emphysema to weather and air pollution. Arch. Environ. Health 16:406-413, 1968.

Carne, S. Air Pollution Study. Proc. Royal Soc. Medicine 57:30-34, 1964.

Carnow, B. W. Sulfur oxides and particles. Effects on health. Proceedings of the National Academy of Science Conference on Health Effects of Air Pollution. U.S. Government Printing Office, Washington, DC. Stock No. 5270-02105, 1973. pp. 263-291.

Carnow, B. W., M. H. Lepper, R. B. Shekelle, and J. Stamler. Chicago air pollution study. Arch. Environ. Health 18:768-776, 1969.

Cassell, E. J., M. D. Lebowitz, and J. R. McCarroll. The Relationship Between Air Pollution, Weather, and Symptoms in an Urban Population. Am. Rev. Res. Dis. 106:677-683, 1972.

Cassell, E. J., M. D. Lebowitz, I. M. Mountain, H. T. Lee, D. J. Thompson, D. W. Wolter, and J. R. McCarroll. Air Pollution, Weather, and Illness in a New York Population. Arch. Environ. Health 18:523-530, 1969.

Cassell, E. J., D. W. Walter, J. D. Mountain, J. R. Diamond, I. M. Mountain, and J. R. McCarroll II. Reconsiderations of mortality as a useful index of the relationship of environmental factors to health. Amer. J. Pub. Health 58:1653-1557, 1968.

Cederlof, R. Urban factor and prevalence of respiratory symptoms and "angina pectoris." Arch. Environ. Health 13:743-748, 1966.

Chappie, M., and L. Lave. The health effects of air pollution. A Reanalysis. J. Urban Economics, 1981.

Chapman, R. S., C. M. Shy, J. F. Finklea, D. E. House, H. E. Goldberg, and C. G. Hayes. Chronic Respiratory Disease in Military Inductees and Parents of School Children. Arch. Environ. Health 27:138, 1973.

Chanska, M. Changes in peak expiratory flow rate (PEFR) in children during three year observations. Ph.D. Dissertation, State Inst. Hygiene, Warsaw, 1980.

Chiaramonte, L. T., J. R. Bongiorno, R. Brown, and M. E. Laano. Air pollution and obstructive respiratory disease in children. NY State J. Med. 70:394, 1970.

Christainsen, G. B. and C. G. Degen. Air pollution and Mortality Rates: A Note on Lave and Seskin's Pooling of Cross-Section and Time-Series Data. J. of Env. Economics and Mang. 7:149-155, 1980.

Ciocco, C. A., and D. J. Thompson. A Follow-up at Donora Ten Years After: Methodology and Findings. Am. J. Publ. Health 51:155-164, 1961.

Clifton, M., D. Kerridge, J. Pemberton, W. Moulds, and J. K. Donoghue. Morbidity and mortality from bronchitis in Sheffield in four periods of severe pollution. In: Proc. 1959 Int. Clean Air Conf. London, National Society for Clean Air. 1960. p. 189.

Cohen, A. A., S. Bromberg, R. W. Buechley, L. T. Heiderescheit, and C. M. Shy. Asthma and air pollution from a coal fueled power plant. Am. Rev. J. Pub. Health. 62, 1972. p. 1181.

Colley, J. R. T., and D. D. Reid. Urban and social origins of childhood bronchitis in England and Wales. Br. Med. J. 2:213-217, 1970.

Colley, J. R. T., and L. J. Brasser. Chronic respiratory diseases in children in relation to air pollution. Report of a WHO study. EVRO Reports and Studies 28, Regional Office for Europe, Copenhagen, 1980.

Colley, J. R. T., and W. W. Holland. Social and Environmental Factors in Respiratory Disease. Arch Environ. Hlth. 14:157, 1967.

Collins, J. J., H. S. Kasap, and W. W. Holland. Environmental factors in child mortality in England and Wales. Am. J. Epidemiol. 93:10, 1971.

Commins, B. T., and R. E. Waller. Observations from a ten-year study of pollution at a site in the city of London. Atmos. Environ. 1:49-68, 1967.

Comstock, G. W., R. W. Stone, Y. Sakai, T. Matsuya, and J. A. Tonascia. Respiratory findings and urban living. Arch. Environ. Health 27:143, 1973.

Cowan, D. W., H. J. Thompson, H. J. Paulus; and P. W. Mielke. Bronchial asthma associated with air pollutants from the grain industry. J. Air Poll. Contr. Assoc. 13:546, 1963.

Crocker, T. G., W. Schulze, S. Ben-David, and A. V. Kneese. Methods development for assessing air pollution control benefits, Volume I: Experiments in the economics of air pollution epidemiology. EPA-600/5-79-001a, Environmental Protection Agency, Research Triangle Park, N.C., 1979.

Deane, M., J. R. Goldsmith, and D. Tuma. Respiratory conditions in outside workers. Report on outside plant telephone workers in San Francisco and Los Angeles. Arch. Environ. Health 10:323, 1965.

Derrick, E. H. A comparison between the density of smoke in the Brisbane air and the prevalence of asthma. Med. J. Aust. 11:670-675, 1970.

Detels, R., J. W. Sayre, A. H. Coulson, S. N. Rokaw, F. J. Massey, Jr., D. P. Tashking and M. Wu. Respiratory Effect of Long Term Exposure to Two Mixes of Air Pollutants in Los Angeles County. Chest 80:27S-29S, 1981.

Dockery, D. W., N. R. Cook, B. G. Ferris, F. E. Speizer, J. D. Spengler, and J. H. Ware. Changes in pulmonary function in children with air pollution episodes. Proceedings 74th Annual APCA, Philadelphia, Pa., June 21-26, 1981.

Dodge, R. The respiratory health of school children in smelter communities. Amer. J. Ind. Med. 1:359-364, 1980.

Dohan, F. C. Air pollutants and incidence of respiratory disease. Arch. Environ. Health 3:387-395, 1961.

Dohan, F. C., and E. W. Taylor. Air Pollution and Respiratory Disease, A Preliminary Report. Am. J. Med. Sci. 240:337, 1960.

Doll, R. Atmospheric pollution and lung cancer. Environ. Health Perspect. 22:23-31, 1978.

Douglas, J. W. B., and R. W. Waller. Air pollution and respiratory infection in children. Br. J. Prev. Soc. Med. 20:1-8, 1966.

Ellison, J. The estimation of particulate air pollution from the soiling of filter paper. Staub Reinhalt, Luft 28:28, 1968.

Emerson, P. A. Air pollution, atmospheric conditions and chronic airway obstructions. J. Occup. Med. 15:635-638, 1973.

Fairbairn, A. S., and D. D. Reid. Air pollution and other local factors in respiratory disease. Br. J. Prev. Soc. Med. 12:94, 1958.

Ferris, B. G., Jr. Health Effects of Exposures to Low Levels of Regulated Air Pollutants: A Critical Review. JAPCA 28:482-497, 1978.

Ferris, B. G., Jr., and D. O. Anderson. The prevalence of chronic respiratory disease in a New Hampshire town. Am. Rev. Respir. Dis. 86:165-177, 1962.

Ferris, B. G., Jr., Burgess, W. A., and J. Worchester, J. Prevalence of chronic respiratory disease in a pulp mill and a paper mill in the United States. Br. J. Ind. Med., 24:26-37, 1967.

Ferris, B. G., Jr., H. Chen, S. Puleo, and R. L. H. Murphy, Jr. Chronic non-specific respiratory disease in Berlin, New Hampshire, 1967-1973. A further follow-up study. Am. Rev. Respir. Dis. 113: 475-485, 1976.

Ferris, B. G., Jr., I. T. T. Higgins, M. W. Higgins, J. M. Peters, W. F. Van Guase, and M. D. Goldman. Chronic non-specific respiratory disease, Berlin, New Hampshire, 1961-67: A cross section study. Am. Rev. Respir. Dis. 104:232-244, 1971.

Ferris, B. G., Jr., I. T. T. Higgins, M. W. Higgins, and J. M. Peters. Chronic non-specific respiratory disease in Berlin, New Hampshire, 1961-67. A follow-up study. Am. Rev. Respir. Dis. 107:110-122, 1973.

Ferris, B. G., Jr., I. T. T. Higgins, M. W. Higgins, and J. M. Peters. Sulfur oxides and suspended particulates, possible effects of chronic exposure. Arch. Environ. Health 27:179-182, 1973.

Ferris, B. G., Jr., J. R. Mahoney, R. M. Patterson, and M. W. First. Air quality, Berlin, New Hampshire, March 1966 to December 1967. Am. Rev. Respir. Dis. 108:77-84, 1973.

Ferris, B. G., Jr., F. E. Speizer, Y. M. M. Bishop, J. D. Spengler, and J. H. Ware. The six-city study: A progress report. In: Atmospheric Sulfur Deposition: Environmental Impact and Health Effects. Ann Arbor Sicence, D. S. Shriner, C. R. Richmond, and S. E. Lindberg, eds., Ann Arbor Science, Ann Arbor, Michigan, 1980. pp. 99-108.

Ferris, B. G., Jr., R. E. Speizer, J. D. Spengler, D. Dockery, Y. M. M. Bishop, M. Wolfson, and C. Humble. Effects of sulfur oxides and respirable particles on human health methodology and demography of populations in study. Am. Rev. Respir. Dis. 120:767-779, 1979.

Firket, J. Sur les causes des accidents survenus dans la valee de la Meuse, lors des brouillards de Decembre, 1930. Bull. Acad. R. Med. Belg. 11:683-741, 1931.

Firket, J. Fog along the Meuse Valley Trans Fraday Soc. 32:1191-1194, 1936.

Fletcher, C. M., R. Peto, C. M. Tinker, and F. E. Speizer. The Natural History of Chronic Bronchitis and Emphysema (an eight year study of early chronic obstructive lung disease in working men in London). Oxford University Press, 1976.

French, J. G., G. Lowrimore, W. C. Nelson, J. F. Finklea, T. English, and M. Hertz. The effect of sulfur dioxide and suspended sulfate on acute respiratory disease. Arch. Environ. Health 27:129-133, 1973.

Fujita, S., T. Motoichi, K. Shoji, Y. Ichiro, F. Takashi, S. Seigo, K. Tatsuo, and M. Michiko. Studies on chronic bronchitis epidemiological survey (2nd report). Teishin Igaku 21:13, 1969.

Gervois, M., G. Dubois, S. Gervois, J. M. Queta, A. Muller, and C. Vorsin. Atmospheric pollution and acute respiratory disease. Denoin and Quavrechoin epidemiological study. Rev. Epidemiol. Sante Publique 25:195-207, 1977.

Gerking, S., and W. Schultze. What do we know about benefits of reduced mortality from air pollution control? Amer. Economic Review 71:228-234, 1981.

Girsh, L. S., E. Shubin, C. Dick, and F. A. Schulaner. A Study on the Epidemiology of Asthma in Children in Philadelphia. J. Allergy 39:347-357, 1967.

Glasser, M., and L. Greenburg. Air pollution and mortality and weather, New York City, 1960-64. Arch. Environ. Health 22:334-343, 1971.

Glasser, M., L. Greenburg, and F. Field. Mortality and Morbidity During a Period of High Levels of Air Pollution, New York, November 23-25, 1966. Arch. Environ. Health 15:684-694, 1967.

Goldsmith, J. R., and L. T. Friberg. Effects of Air Pollution on Human Health. In: Air Pollution . II. A. C. Stern, ed., Academic Press, New York, 1977. pp. 458-610.

Goldstein, I. F., and G. Block. Asthma and air pollution in two inner city areas in New York City. J. Air Pollut. Control Assoc. 24:665-670, 1974.

Goldstein, I. F. and E. M. Dulberg. Asthma and air pollution in two inner city areas in New York City. J. Air Pollut. Contro. Assoc. 24:370-376, 1981.

Goldstein, I., and L. Landowitz (Letter to editor). J. Air Pollut. Control Assoc. 25:1195, 1975.

Goldstein, I. F. and L. Landovitz. Analysis of air pollution patterns in New York City--I. Can one station represent the large metropolitan area? Atmos. Environ. 11:47-52, 1977a.

Goldstein, I. F. and L. Landovitz. Analysis of air pollution patterns in New York City--11. Can one aerometric station represent the area surrounding it? Atmos. Environ. 11:53-57, 1977b.

Gore, A. T., and C. W. Shaddick. Atmospheric pollution and mortality in the County of London. Br. J. Prev. Soc. Med. 12:104-113, 1958.

Gorham, E. Bronchitis and the acidity of urban precipitation. Lancet 2:691, 1958.

Gorham, E. Pneumonia and atmospheric sulphate deposit. Lancet 2:287, 1959.

Greenberg, L., M. Jacobs, B. Drolette, F. Field, and M. Braverman. Report of an air pollution incident in New York City, November, 1953. Pub. Health Rep. 77:7-16, 1962.

Greenburg, L., C. Erhardt, F. Field, J. I. Reid, and N. S. Seriff. Intermittent air pollution episode in New York City, 1962. Public Health Rep. 78:1061-1064, 1963.

Greenburg, L., F. F. Field, J. I. Reed, and C. L. Erhardt. Air pollution and morbidity in New York City. J. Am. Med. Assoc. 182:161-164, 1962.

Greenburg, L., F. Field, J. I. Reed, and C. L. Erhardt. Asthma and temperature change. An epidemiological study of emergency clinic visits for asthma in three large New York Hospitals. Arch. Environ. Health 8:642, 1964.

Greenburg, L., F. Field, C. Erhardt, M. Glasser, and J. Reed. Air pollution, influenza, and mortality in New York City. Arch. Environ. Health 15:430, 1967.

Gregory, J. The influence of climate and atmospheric pollution on exacerbations of chronic bronchitis. Atmos. Environ. 4:453-468, 1970.

Hagström, R. M., H. A. Sprague, and E. Landau. The Nashville air pollution study. VII. Mortality from cancer in relation to air pollution. Arch. Environ. Health 15:237-248, 1967.

Hammer, D. I., F. J. Miller, A. G. Stead, and C. G. Hayes. Air Pollution and Childhood Lower Respiratory Disease. I. Exposure to Sulfur Oxides and Particulate Matter in New York, 1972. In: Clinical Implications of Air Pollution Research. A. J. Finkel and W. C. Duel, ed., Publishing Sciences Group, Inc., Acton, MA, 1976. pp. 321-337.

Hammer, D. I. Frequency of lower respiratory disease in children: Retrospective survey of two southeastern communities, 1968-1971. Ph.D. Dissertation, Harvard, Univ., 1976.

Heimann, H. Episodic air pollution in metropolitan Boston. A trial epidemiologic study. Arch. Environ. Health 20:230-251, 1970.

Hewitt, D. Mortality in the London boroughs, 1950-52, with special reference to respiratory disease. Br. J. Prev. Soc. Med. 10:45, 1956.

Higgins, I. T. T. Epidemiology of Chronic Respiratory Disease: A Literature Review. EPA-650/1-74-007, U.S. Environmental Protection Agency, DC, 1974.

Higgins, I. T. T. Trends in respiratory cancer mortality in the U.S. and in England and Wales. Arch. Environ. Health. 28:121-129, 1976.

Hill, A. B. The Environment and Diseases: Associations and Causation. In: Proceedings of the Royal Society of Medicine (Occ. Med.) 58:272, 1965.

Hodgson, A., Jr. Short-term effects of air pollution on mortality in New York City. Environ. Sci. Technol. 4:589-597, 1970.

Holland, W. W., A. E. Bennett, I. R. Cameron, C. du V. Florey, S. R. Leeder, R. S. F. Schilling, A. V. Swan, and R. E. Waller. Health Effects of Particulate Pollution: Re-appraising the Evidence. Am. J. Epidemiol. 110(5):525-659, 1979.

Holland, W. W., and D. D. Reid. The Urban Factor in Chronic Bronchitis. Lancet 1:445-448, 1965.

Holland, W. W., and R. W. Stone. Respiratory disorders in United States East Coast telephone men. Am. J. Epidemiol. 82:92-101, 1965.

Holland, W. W., ed. Data Handling in Epidemiology. Oxford University Press, London, 1970.

Holland, W. W., T. Halil, A. E. Bennett, and A. Elliot. Indications for measures to be taken in childhood to prevent chronic respiratory disease. Milbank Mem. Fund Q. 47:215-227, 1969a.

Holland, W. W., T. Halil, A. E. Bennett, and A. Elliot. Factors influencing the onset of chronic respiratory disease. Br. Med. J. 2:205-208, 1969b.

Holland, W. W., H. S. Kasap, J. R. T. Colley, and W. Cormack. Respiratory symptoms and ventilatory function: A family study. Br. J. Prev. Soc. Med. 23:77-84, 1969c.

Hosein, H. R., C. A. Mitchell, and A. Bouhuys. Evaluation of outdoor air quality in rural and urban communities. Archives of Environ. Health. Jan./Feb. 4-13, 1977a.

Hosein, H. R., Mitchell, C. A., and A. Bouhuys. Daily variation in air quality. Arch. of Environ. Health Jan./Feb. 14-21, 1977b.

Hrubec, Z., R. Cederlof, L. Freberg, R. Horton, and G. Ozolins. Pespiratory symptoms in twins. Arch. Environ. Health 27:189-195, 1973.

Ingram, W. Smoke Curve Calibration. PHS Contract PH-86-68-66, New York University, New York, NY, 1969.

Ingram, W. T., and J. Golden. Smoke curve calibration. J. Air Pollut. Control Assoc. 23:110, 1973.

Ipsen, J., M. Deane, and F. E. Ingenito. Relationship of acute respiratory disease to atmospheric pollution and meteorological condition. Arch. Environ. Health 18:462-472, 1969.

Irwig, L., D. G. Altman, R. J. W. Gibson, and C. Du V. Florey. Air Pollution: Methods to study its Relationship to Respiratory Disease in British Schoolchildren. Proceedings of the Intermath Symp on Recent Advances with Asses. of the Health Effects of Environ. Pol., Luxembourg: Commission of the European Communities, Vol 1, 1975 pp. 289-300.

Ishikawa, S., D. H. Bowden, V. Fisher, and J. P. Wyatt. The "emphysema profile" in two mid-western cities in North America. Arch. Environ. Health 18:660, 1969.

Jacobs, C., and B. Langdoc. Cardiovascular deaths and air pollution in Charleston, South Carolina. Health Services Reports 87:623-632, 1972.

Johnson, K. G., R. Gideon, and D. Loftsgaonden. Montana air pollution study. Synopsis of pulmonary function studies. Proceedings of 74th Annual APCA Meeting, Philadelphia, Pa., June 21-26, 1981.

Kagawa, J., and T. Toyama. Photochemical Air Pollution. Arch. Environ. Health 30:117-122, 1975.

Kagawa, J., T. Toyama, and M. Nakaza. Pulmonary function tests in children exposed to air pollution. In: Clinical Implications of Air Pollution Research. A. J. Finkel and W. C. Duel, ed., Publishing Sciences Group, Inc., Acton, MA, 1976. pp. 305-320.

Kalpazanov, Y., M. Stamenova, and G. Kurchatova. Air pollution and the 1974-1975 influenza epidemic in Sofia. Environ. Res. 12:1-8, 1976.

Kenline, P. In quest of clean air for Berlin, New Hampshire, U.S. Department of Health, Education, and Welfare. R. A. Taft Sanitary Engineering Center, Cincinnati, Ohio, 1962.

Kenline, P. A. October 1963 New Orleans asthma study. Arch. Environ. Health 12:295-304, 1966.

Kevany, J., M. Rooney, and J. Kennedy. Health effects of air pollution in Dublin. Ir. J. Med. Sci. 144:102-115, 1975.

Kiernan, K. E., J. R. T. Colley, J. W. B. Douglas, and D. D. Reid. Chronic cough in young adults in relation to smoking habits, childhood environment and chest illness. Respiration 33:236-244, 1976.

Lambert, P. M., and D. D. Reid. Smoking, air pollution and bronchitis in Britain. Lancet i:853-857, 1970.

Lave, L. B., and B. P. Seskin. Air pollution and human health. The quantitative effect, with an estimate of the dollar benefit of pollution abatement is considered. Science 169:723-733, 1970.

Lave, L. B., and B. P. Seskin. Air pollution, climate, and home heating: Their effects on U.S. mortality rate. Am. J. Public Health 62:909-916, 1972.

Lave, L. B., and B. P. Seskin. Air Pollution and Human Health. Baltimore, The Johns Hopkins University Press. 1977.

Lawther, P. J. Climate, air pollution and chronic bronchitis. Proc. R. Soc. Med. 51:262-264, 1958.

Lawther, P. J. Compliance with the Clean Air Act: Medical aspects. J. Inst. Fuel 36:341, 1963.

Lawther, P. J., A. G. F. Brooks, P. W. Lord, and R. E. Waller. Day-to-day changes in ventilatory function in relation to the environment. Part I. Spirometric values. Environ. Res. 7:24-40, 1974a.

Lawther, P. J., A. G. F. Brooks, P. W. Lord, and R. E. Waller. Day-to-day changes in ventilatory function in relation to the environment. Part II. Peak expiratory flow values. Environ. Res. 7:41-53, 1974b.

Lawther, P. J., A. G. F. Brooks, P. W. Lord, and R. E. Weller. Day-to-day changes in ventilatory function in relation to the environment. Part III. Frequent measurements of peak flow. Environ. Res. 8:119-130, 1974c.

Lawther, P. J., P. W. Lord, A. G. F. Brooks, and R. E. Waller. Air pollution and pulmonary airway resistance: A pilot study. Environ. Res. 6:424-435, 1973.

Lawther, P. J., R. E. Waller, and M. Henderson. Air pollution and exacerbations of bronchitis. Thorax 25:525-539, 1970.

Lebowitz, M. D. A comparative analysis of the stimulus-response relationship between mortality and air pollution weather. Environ. Res. 6:106-118, 1973a.

Lebowitz, M. D., T. Toyama, and J. McCarrol. The relationship between air pollution and weather as stimuli and daily mortality as responses in Tokyo, Japan, with comparisons with other cities. Environ. Res. 6:327-333, 1973b.

Lebowitz, M. D., E. J. Cassell, and J. D. McCarroll. Health and the Urban Environment. XV. Acute Respiratory Episodes as Reactions by Sensitive Individuals to Air Pollution and Weather. Environ. Research 5(2):135-141, 1972.

Lebowitz, M., P. Bendheim, G. Cristea, D. Markovitz, J. Misiaszek, M. Staniec, and D. Van Wyck. The effect of air pollution and weather on lung function in exercising children and adolescents. Am. Rev. Respir. Dis. 109:262-273, 1974.

Lee, R. E. Jr., J. S. Caldwell, and G. B. Morgan. The evaluation of methods for measuring suspended particulates in air. Atmos. Environ. 6:593-622, 1972.

Lepper, M. H., N. Shioura, B. Carnow, S. Andelman, and L. Lehrer. Respiratory disease in an urban environment. Arch. Indust. Med. 38:36, 1969.

Levy, D., M. Gent, and M. T. Newhouse. Relationship between acute respiratory illness and air pollution levels in an industrial city. Am. Rev. Respir. Dis. 116:167-175, 1977.

Lindeberg, W. Correlations between air pollutant concentrations and death rates in Oslo. In: Air Pollution in Norway. III. Oslo, Norway, Smoke Damage Council, 1968.

Linn, W. S., J. D. Hackney, E. E. Pedersen, P. Breisacher, J. V. Patterson, C. A. Mulry, and J. F. Coyle. Respiratory function and symptoms in urban office workers in relation to oxidant air pollution exposure. Am. Rev. Res. Dis. 114:477, 1976.

Lipfert, F. W. Sulfur oxides, particulates, and human mortality: Synopsis of statistical correlations. J. Air Pollut. Control Assoc. 30:366-371, 1980.

Logan, W. Mortality in the London fog incident. Lancet 1:336-338, 1953.

Lowrence, W. W. Of Acceptable Risk. Science and Determination of Safety. Los Altos, William Kaufman, 1976.

Liu, B. Y. H., D. Y. H. Pui, K. L. Rubow, and G. A. Kuhlmey. Progress Report--Research on Air Sampling Filter Media. Grant Report R804600, University of Minnesota, Minneapolis, MN, May 1978.

Lunn, J. E., J. Knowelden, and A. J. Handyside. Patterns of respiratory illness in Sheffield infant schoolchildren. Br. J. Prev. Soc. Med. 21:7-16, 1967.

Lunn, J. E., J. Knowelden, and J. W. Roe. Patterns of respiratory illness in Sheffield junior schoolchildren. A follow-up study. Br. J. Prev. Soc. Med. 24:223-228, 1970.

Manfreda, J., N. Nelson, and R. M. Cherniack. Prevalence of respiratory abnormalities in a rural and an urban community. Am. Rev. Respir. Dis. 117:215-226, 1978.

Martin, A. E. Mortality and morbidity statistics and air pollution. Proc. R. Soc. Med. 57:969-975, 1964.

Martin, A. E., and W. H. Bradley. Mortality, fog and atmospheric pollution--An investigation during the winter of 1958-59. Mon. Bull. Minist. Health Public Health Lab. Serv. 19:56-72, 1960.

Mazumdar, S., and N. Sussman. Relationships of air pollution to health: Results from the Pittsburgh Study. Proceedings of 74th Annual APCA Meeting, Philadelphia, Pa., June 21-26, 1981.

Mazumdar, S., H. Schimmel, and I. Higgins. Daily mortality, smoke, and SO_2 in London, England, 1959-1972. Proceedings of APCA Conference on the Proposed SO_2 and Particulate Standard, Sept. 16-18, 1980, Pittsburgh, Pa., 1981. pp. 219-239.

McCarroll, J. R., and W. H. Bradley. Excess mortality as an indicator of health effects of air pollution. Am. J. Pub. Health 56:1933, 1966.

McCarroll, J. R., E. J. Cassell, W. T. Ingram, and D. Wolter. I. Health and the Urban Environment. Am. J. Public Health 56:266-275, 1966.

McCarroll, J., E. J. Cassell, D. W. Woeter, J. D. Mountain, J. R. Diamond, and I. M. Mountain. Health and the Urban Environment. Arch. Environ Health 14:178-1967.

McFarland, A. R., C. A. Ortiz, and C. E. Rodes. Wind tunnel evaluation of the British smoke shade sampler. Atmos. Environ. 16:325-328, 1982.

McFarland, A. R., and C. A. Ortiz. Aerosol characterization of ambient particulate samplers used in environmental monitoring. Progress Report, EPA Contract 68-02-2720, Air Quality Laboratory, Texas A & M University, College Station, Texas, 1980.

Mendelsohn, R. and G. Orcutt. An Empirical Analysis of Air Pollution Dose-Response Curves. J. of Env. Economics and Mang. 6:85-106, 1979.

Ministry of Health. Mortality and Morbidity During the London Fog of December 1952. London, Her Majesty's Stationery Office. 1954.

Ministry of Pensions and National Insurance. Report on an Enquiry into the Incidence of Incapacity for Work. II. Incidence of Incapacity for Work in Different Areas and Occupations. London, Her Majesty's Stationery Office, 1965.

Mitchell, C. A., R. S. F. Schilling, and A. Bouhuys. Community studies of lung disease in Connecticut: organization and methods. Am. J. of Epi. 103:213-224, 1976.

Mork, T. A comparative study of respiratory disease in England , Wales, and Norway. Norwegian University Press, Oslo, 1962.

Morris, S. C., M. A. Shapiro, and J. H. Waller. Adult mortality in two communities with widely different air pollution levels. Arch. Environ. Health 31:248-254, 1976.

Mostardi, R. A., D. L. Ely, N. R. Woebkenberg, and M. Conlon. Air Pollution and Health Effects in Children Residing in Akron, Ohio. Chest 80:265-275, 1981.

Mostardi, R. A. and D. Leonard. Air Pollution and Cardiopulmonary Functions. Arch. Environ. Health 29:325-328, 1974.

Mostardi, R. A. and R. Martell. The Effects of Air Pollution on Pulmonary Functions in Adolescents. The Ohio J. of Science. 75:65-69, 1975.

Moulds, W. Some instrumental variations arising in routine air pollution measurements. Int. J. Air Water Pollut. 6:201-203, 1962.

Mountain, I. M., E. J. Cassell, D. W. Wolter, and J. D. Mountain. Health and the Urban Environment. VII. Air Pollution and Disease Symptoms in a Normal Population. Arch. Environ. Health 17:343-352, 1968.

NAS. Proceedings of the Conference on Health Effects of Air Pollutants, prepared for the Committee on Public Works, U.S. Senate, Committee Print, Serial no. 93-15, U.S. Government Printing Office, Washington, DC, 1978.

National Research Council. Airborne Particles. National Academy of Sciences. Washington, DC, 1978, Chapter 9, Epidemiological Studies on the Effects of Airborne Particles on Human Health. I. T. T. Higgins and B. G. Ferris, Jr. pp. 243-288, 1978a.

National Research Council. Sulfur oxides. National Academy of Sciences. Washington, DC, 1978, Chapter 7. Epidemiological Studies of Health Effects. F. E. Speizer and B. G. Ferris, Jr.. pp. 180-209, 1978b.

Neri, L. C., J. S. Mandel, D. Hewitt, and D. Jurkowski. Chronic obstructive pulmonary disease in two cities of contrasting air quality. Can. Med. Assoc. J. 113:1043-1046, 1975.

Nobuhiro, T., M. Yozo, T. Yoshizo, K. Kiroyuri, H. Masamichi, K. Tachachiro, H. Teruo, and H. Ken'ichi. Concerning air pollution and chronic bronchitis in Ako City. Report of the Environment Pollution Research Institute of Hyogo Prefecture, Japan. 1:25-35, 1970.

Organization for Economic Co-operation and Development. Methods of Measuring Air Pollution. Paris, France, 1965.

Pedace, E. A., and E. B. Sansone. The relationship between "soiling index" and suspended particulate matter concentrations. J. Air Pollut. Control Assoc. 22:348-351, 1972.

Pemberton, J., and C. Goldberg. Air polution and bronchitis. Br. Med. J. 2:557, 1954.

Petrilli, F. L., G. Agnese, and S. Kanitz. Epidemiologic studies of air pollution effects in Genoa, Italy. Arch. Environ. Health 12:733-740, 1966.

Prindle, R. A., G. W. Wright, R. O. McCaldin, S. C. Marcus, T. C. Lloyd, and W. E. Bye. Comparison of pulmonary function and other parameters in two communities with widely different air pollution levels. Am. J. Public Health 53:200, 1963.

Rall, D. P. A Review of the Health Effects of Sulfur Oxides. National Institute of Environmental Health Sciences, NIH, Research Triangle Park, NC, 1973, Environ. Hlth. Perspect. 8:97-121, 1974.

Ramaciotti, D., M. Bahy, B. Voinier, and P. Rey. The SO_2 pollution level and the incidence of bronchitis. Medicine sociale et preventive 22:189-190, 1977.

Ramsey, J. M. The relationship of urban atmospheric variables to asthmatic bronchoconstriction. Bull. Environ. Contam. Toxicol. 16:107-111, 1976.

Rao, M., P. Steiner, Q. Qazi, R. Padre, J. E. Allen, and M. Steiner. Relationship of air pollution to attack rate of asthma in children. J. Asthma Res. 11:23, 1973.

Reichel, G. Effect of air pollution on the prevalence of respiratory diseases in West Germany. In: Proceedings of the Second International Clean Air Congress, Washington, DC, 1970.

Rudnik, J. Epidemiological Study on Long-term Effects on Health of Air Pollution. Probl. Med. Wieku Rozwojowego 7a(suppl):1-159, 1978.

Samet, J. M., Y. Bishop, F. E. Speizer, J. D. Spengler, and B. G. Ferris, Jr. The Relationship Between Air Pollution and Emergency Room Visits in an Industrial Community. JAPCA 31: 236-240, 1981.

Saric, M., M. Fugas, and O. Hrustic. Effects of Urban Air Pollution on School-Age Children. Arch. of Environ. Health, 36:101-108, 1981.

Sawicki, F. Air pollution and prevalence of non-specific chronic respiratory disease. In: Ecology of Chronic Non-Specific Respiratory Diseases. Z. Brzezinski, J. Kopczynski, and F. Sawicki. ed., Warsaw, Panstwowy Zaklad Wydawnictw Lekarskich. 1972. p. 3-13.

Sawicki, F. Chronic non-specific respiratory disease in the city of Cracow. X. Statistical analysis of air pollution by suspended particulate matter and sulfur dioxide. Epidemiol. Rev. 23:221, 1969.

Sawicki, F. Chronic non-specific respiratory disease in the city of Cracow. XI. The cross-section study. Epidemiol. Rev. 23:242, 1969.

Sawicki, F., and P. S. Lawrence, eds. Chronic Non-specific Respiratory Disease in the City of Cracow--Report of a 5 year Follow-up Study Among Adult Inhabitants of the City of Cracow. National Institute of Hygiene, Warsaw, Poland, 1977.

Schimmel, H. Evidence for possible acute health effects of ambient air pollution from time series analysis-methodology questions and some new results based on New York City daily mortality, 1963-1976. Bull, N.Y. Acad. 54:1052-1108, 1978.

Schimmel, H., and T. J. Murawski. SO2--Harmful pollutant or air quality indicator? J. Air Pollut. Control Assoc. 25:739-740, 1975.

Schimmel, H., and T. J. Murawski. The relation of air pollution to mortality. J. Occup. Med. 18:316-333, 1976.

Schimmel, N., and L. Greenburg. A study of the relationship of pollution to mortality, New York City, 1963-1968. J. Air Pollut. Control Assoc. 22:607-616, 1972.

Schoettlin, C. E., and E. Landau. Air pollution and asthmatic attacks in the Los Angeles area. Public Health Reports 76:545, 1961.

Schrenk, H. H., H. Heimann, G. D. Clayton, W. Gafafer, and H. Wexler. Air Pollution in Donora, Pennsylvania. Epidemiology of the Unusual Smog Episode of October 1948. Public Health Bulletin 306, U.S.G.P.O. Washington, DC, 1949.

Schwing, R. C. and G. G. McDonald. Measures of Association of Some Air Pollutants, Natural Ionizing Radiation and Cigarette Smoking with Mortality Rates. The Science of the Total Environment 5:139-169, 1976.

Scott, J. A. Fog and deaths in London, December 1952. Pub. Health Rep. 68:474-479, 1953.

Scott, J. A. The London fog of December, 1962. Med. Off. 109: 250-252, 1963.

Shy, C. M. Epidemiologic Evidence and the United States Air Quality Standards. Am. J. Epidemiol. 110:661-671, 1979.

Shy, C. M., J. R. Goldsmith, J. D. Hackney, M. D. Lebowitz, and D. B. Menzel. Health Effects of Air Pollution. American Thoracic Society, Medical Section of American Lung Association, 1978.

Shy, C. M., V. Hasselblad, R. M. Burton, C. J. Nelson, and A. Cohen. Air Pollution Effects on Ventilatory Function of U.S. Schoolchildren. Results of Studies in Cincinnati, Chattanooga, and New York. Arch. Environ. Health 27:124-128, 1973.

Smith, T. J., and H. J. Paulus. An Epidemiology Study of Atmospheric Pollution and Bronchial Asthma Attacks. Presented at the 64th Annual Meeting of the Air Pollution Control Assoc., Atlantic City, June 27-July 1, 1971. 44 pp.

Speizer, F. E. An Epidemiological Appraisal of the Effects of Ambient Air on Health: Particulates and Oxides of Sulfur. J. Air Pollut. Control Assoc. 19:647-655, 1969.

Speizer, F. E., and B. G. Ferris, Jr. Exposure to automobile exhaust. I. Prevalence of respiratory symptoms and disease. Arch. Environ. Health 26:313, 1973a.

Speizer, F. E., and B. G. Ferris, Jr. Exposure to automobile exhaust. II. Pulmonary function measurements. Arch. Environ. Health 26:319, 1973b.

Sprague, H. A., and R. M. Hagstrom. The Nashville air pollution study: Mortality multiple regression. Arch. Environ. Health 18:503-507, 1969.

Stebbings, J. H., Jr., and D. G. Fogleman. Identifying a Susceptible Subgroup: Effects of the Pittsburgh Air Pollution Episode Upon Schoolchildren. Am. J. Epidemiol. 110:27-40, 1979.

Stebbings, J., and C. Hayes. Panel Studies of acute health effects of air pollution. I. Cardiopulmonary symptoms in adults, New York, 1971-1972. Environ. Res. 11:89-111, 1976.

Sterling, J. D., J. J. Phair, S. V. Pollack, D. A. Schumsky, and I. De Grout. Urban Morbidity and Air Pollution. A First Report. Arch. Environ. Health 13:158-1966.

Sterling, J. D., S. V. Pollack, and J. J. Phair. Urban Hospital Morbidity and Air Pollution. A Second Report. Arch Environ Health 15:352-1967.

Stocks, P. Air Pollution and Cancer Mortality in Liverpool Hospital Region and North Walls. Inter. J. Air Pollut. 1:1-13, 1958.

Stocks, P. Cancer and bronchitis mortality in relation to atmospheric deposit and smoke. Br. Med. J. 1:74, 1959.

Stocks, P. On the relations between atmospheric pollution in urban and rural localities and mortality from cancer, bronchitis and pneumonia with particular reference to 3:4-benzopyrene, beryllium, molybdenum, vanadium, and arsenic. Br. J. Cancer 14:397-418, 1960a.

Stocks, P., and R. I. Davies. Epidemiological evidence from chemical and spectrographic analyses that soil is concerned in the causation of cancer. Br. J. Cancer 14:8-22, 1960b.

Sultz, H., J. Feldman, E. Schlesinger, and W. Mosher. An effect of continued exposure to air pollution on the incidence of chronic childhood allergic disease. Am. J. Public Health 60:891-900, 1970.

Suzuki, T., N. Ishinishi, R. Yoshida, Y. Tsunetoshi, M. Hitosugi, S. Tominaga, K. Fukutomi, and A. Nozoe. The Relationship Between Air Pollution and the Respiratory Symptoms and Functions of Housewives. Japan Public Health Society Foundation, Tokyo, Japan, 1978.

Thibodeau, L. A., R. B. Reed, Y. M. M. Bishop, and L. A. Kammerman. Air Pollution and Human Health: A Review and Reanalysis. Environmental Health Perspecitives 34:165-183, 1980.

Thompson, D. J., M. D. Lebowitz, E. J. Cassell, D. Wolter, and J. McCarroll. Health and the Urban Environment. VIII. Air Pollution, Weather, and the Common Cold. Am. J. Public Health 60(4):731-739, 1970.

Toyama, T. Air pollution and its health effects in Japan. Arch. Environ. Health 8:153-173, 1964.

Toyama, T., H. Kanyo, K. Makamura, J. Kagawa, S. Yakura, S. Adachi, N. Yamoto, F. Iriyama, F. Kumagaya, S. Osawa, and T. Nakamura. Study on the prevalence of respiratory symptoms in a rural area (Kashima, Ibaragi Pref.) in Japan. J. Jpn. Soc. Air Pollut. 7:24-35 (in Japanese), 1966.

Tsunetoshi, Y., T. Shimizu, H. Takahashi, A. Schenosowa, M. Ueda, N. Nakayama, Y. Yamagata, and A. Ohshino. Epidemiologic study of chronic bronchitis with special reference to effects of air pollution. Int. Arch. Arbeitsmed. 29:1-27, 1971.

U.S. Congress. Committee on Public Works, U.S. Government Printing Office, Washington, DC, 1968. Air Quality Criteria Staff Report, 90th Congress, 2d Session, 1968.

U.S. Department of Health, Education and Welfare. Air Quality Criteria for Sulfur Oxides. Washington, D.C., U.S. Government Printing Office. 1970. 178 p. National Air Pollution Control Administration Publication No. AP-50.

U.S. Department of Health, Education, and Welfare. Air Quality Criteria for Particulate Matter. Washington, D.C., U.S. Government Printing Office, 1970. 211 p. National Air Pollution Control Administration Publication No. AP-49.

U.S. Environmental Protection Agency. Health Consequences of Sulfur Oxides: A Report from CHESS, 1970-71. EPA-650/1-74-004. May 1974.

U.S. Environmental Protection Agency. Scientific and Technical Issues Relating to Sulfates. Ad Hoc Panel of the Science Advisory Board., Washington, DC, 1975.

U.S. House of Representatives. Committee on Science and Technology. The Environmental Protection Agency's Research Program with Primary Emphasis on the Community Health and Environmental Surveillance System (CHESS): an Investigative Report. Government Printing Office, Washington, DC, November 1976.

U.S. Surgeon General's Advisory Comm. on Smoking and Health, 1964.

Ulmer, W. T., G. Reichel, A. Czeike, and A. Leuschner. Regional incidence of nonspecific respiratory diseases. IV. Communication, Int. Arch. Arbeitsmed. 27:73, 1970.

Van der Lende, R. Epidemiology of Chronic Non-Specific Lung Disease (Chronic Bronchitis). Assen, Royal Van Gorcum, and Springfield, Ill., Charles C. Thomas. 1969.

Van der Lende, R., C. Huygen, E. J. Jansen-Koster, S. Knijpstra, R. Peset, B. F. Visser, E. H. E. Wolfs, and N. G. M. Orie. A temporary decrease in ventilatory function of an urban population during an acute increase in air pollution. Bull. Physiopathol. Respir. 11:31-43, 1975.

Van der Lende, R., G. J. Temmeling, B. F. Visser, K. de Vries, J. Wever-Hess, and N. G. M. Orie. Epidemiological investigations in the Netherlands into the influence of smoking and atmospheric pollution on respiratory symptoms and lung function disturbances. Pneumonologie 149:119-126, 1973.

Van der Lende, R., J. P. M. de Kroon, G. J. Tammeling, B. F. Visser, K. de Vries, J. Wever-Hess, and N. G. M. Orie. Prevalence of chronic non-specific lung disease in a non-polluted and an air polluted area of the Netherlands. In: Ecology of Chronic Non-Specific Respiratory Diseases. Z. Brzezinski, J. Kopczynski, and F. Sawicki, ed., Warsaw, Panstwowy Zaklad Wydownictw Lekarskick. 1972. p. 27-33.

Van der Lende, R., T. J. Kok, R. P. Reig, J. P. Schouten, N. G. M. Orie, and P. H. Quanjer. The relationship between air pollution and diminished lung function values. Results of a longitudinal epidemiological study in the Netherlands. Proceedings of 74th Annual APCA Meeting, Philadelphia, Pa., June 21-26, 1981.

Verma, M. P., F. J. Schilling and W. H. Becker. Epidemiological Study of Illness Absences in Relation to Air Pollution. Arch Environ Health 18:536-543, 1969.

Waller, R. E. Experiments on the calibration of smoke filters. J. Air Pollution Control Association, 14:323-335, 1964.

Waller, R. E., A. G. F. Brooks, and J. Cartwright. An electron microscope study of particles in town air. Int. J. Water Pollut. 7:779-786, 1963.

Waller R. E., A. G. F. Brooks, and M. W. Adler. Respiratory Symptoms and ventilatory capacity in a cohort of Londoners born in 1952-53. Proceedings of the International Symposium on recent advances in the assessment of the health effects of environmental pollution, Paris, June 1974.

Waller, R. E. Health Effects of Exposure to Low Levels of Regulated Air Pollutants. J. Air Pollut. Control Assoc. 28:884-887, 1978.

Waller, R. E., and P. J. Lawther. Some observations on London fog. Br. Med. J. 1:1356-1358, 1955.

Ware, J. H., L. A. Thibodeau, F. E. Speizer, S. Colome, and B. G. Ferris, Jr. Assessment of the Health Effects of Atmospheric Sulfur Oxides and Particulate Matter: Evidence from Observational Studies. Environ. Health Perspect. 41: 255-276, 1981.

Warren Spring Laboratory. Accuracy and representativeness of the National Survey data. In: National Survey of Air Pollution, 1961-1971. Volume 5. Scotland, Northern Ireland, Accuracy of data, Index. Warren Spring Labotory, Stevenage, England, January 1975. pp. 111-119.

Warren Spring Laboratory. Measurement of Atmospheric Smoke and Sulphur Dioxide: Reproducibility of Results. RR/AP/70, Warren Spring Laboratory, Stevenage, England, August 1962.

Warren Spring Laboratory. National Survey of Smoke and Sulphur Dioxide: Instruction Manual. Warren Spring Laboratory, Stevenage, England, 1966.

Warren Spring Laboratory. The Investigation of Atmospheric Pollution 1958-1966. Thirty-second Report. Her Majesty's Stationary Office, London, England, 1967.

Warren Spring Laboratory. The National Survey of Air Pollution. The Use of the Daily Instrument for Measuring Smoke and Sulphur Dioxide. Warren Spring Laboratory, Stevenage, England, December 1961.

Warren Spring Laboratory. The National Survey of Smoke and Sulphur Dioxide-Quality Control Tests on Analyses of Samples, October 1975 to February 1977. Warren Spring Laboratory, Stevenage, England, 1977.

Watanabe, H. Air pollution and its health effects in Osaka. Presented at the 58th Annual Meeting of Air Pollution Control Association, Toronto, Canada, June 20-24, 1965.

Watanabe, H. Health effects of air pollution in Osaka City. J. Osaka Life Hyg. Assoc. 10:147-157(in Japanese), 1966.

Watanabe, H., and F. Kaneko. Excess death study of air pollution. In: Proceedings of the Second International Clean Air Congress. H. M. Englund and W. T. Beery, ed., Academic Press, New York, 1971. pp. 199-200.

Wedding, J. B., A. R. McFarland, and J. E. Cermak. Large particle collection characteristics of ambient aerosol samplers. Environ. Sci. and Technol. 11:389-390, 1977.

Whittemore, A. S., and E. L. Korn. Asthma and air pollution in the Los Angeles area. Am. J. of Pub. Health 70:687-696, 1980.

WHO. Environmental Health Criteria (8): Sulfur Oxides and Suspended Particulate Matter. World Health Organization, Geneva, 1979.

Wicken, A. J., and S. F. Buck. Report on a study of environmental factors associated with lung cancer and bronchitis deaths in areas of northeast England. Research Paper No. 8. London, Tobacco Research Council. 1964.

Wilkins, E. Air pollution and the London Fog of December, 1952. J. Roy. Sanit. Inst. 64:1-21, 1954a.

Wilkins, E. Air pollution aspects of the London Fog of December, 1952. Roy. Meterol. Soc. J. 80:267-271, 1954b.

Winkelstein, W. Utility or futility of ordinary mortality statistics in the study of air pollution effects. In: Proceedings of the Sixth Berkeley Symposium on Mathematical Statistics and Probability. L. LeCam, J. Newyman, and E. Scott, eds., University of California Press, Berkeley, CA, 1972. pp. 539-554.

Winkelstein, W., and M. Gay. Suspended particulate air pollution. Relationship to mortality from cirrhosis of the liver. Arch. Environ. Health 22:174-177, 1971.

Winkelstein, W., and S. Kantor. Stomach cancer. Arch. Environ. Health 14:544-547, 1967.

Winkelstein, W., Jr., and S. Kantor. Respiratory symptoms and air pollution in an urban population of northeastern United States. Arch. Environ. Health 18:760, 1969.

Winkelstein, W., S. Kantor, E. Davis, C. Maneri, and W. Mosher. The relationship of air pollution and economic status to total mortality and selected respiratory system mortality in men. I. Suspended particulates. Arch. Environ. Health 14:162-170, 1967.

Winkelstein, W., S. Kantor, E. Davis, C. Maneri, and W. Mosher. The relationship of air pollution and economic status to total mortality and selected respiratory system mortality in man. II. Oxides of Sulfur. Arch. Environ. Health. 15:401-405, 1968.

Yoshida, R., K. Motomiya, H. Saito, and S. Funabashi. Clinical and epidemiological studies on childhood asthma in air polluted areas in Japan. In: Clinical Implications of Air Pollution Research. Acton, Massachusetts, Publishing Sciences Group, Inc., 1976.

Yoshii, M., J. Nonoyama, H. Oshima, H. Yamagiwa, and S. Taked. Chronic pharyngitis in air-polluted districts of Yo KKAICHI in Japan. Mie Med. J. 19:17-27, 1969.

Zapletal, A., J. Jech, T. Paul, and M. Samanek. Pulmonary function studies in children living in an air polluted area. Am. Rev. Respir. Dis. 107:400-409, 1973.

Zagraniski, R. T., B. P. Leaderer, and J. A. J. Stolwijk. Ambient sulfates, photochemical oxidants, and acute adverse health effects: an epidemiological study. Environ. Res. 19:306-320, 1979.

Zeidberg, L. D., R. A. Prindle, and E. Landau. The Nashville air pollution study. I. Sulfur dioxide and bronchial asthma. A preliminary report. Am. Rev. Res. Dis. 84:489, 1961.

Zeidberg, L. D., R. J. M. Horton, and E. Landau. The Nashville air pollution study. Mortality and diseases of the respiratory system in relation to air pollution. Arch. Environ. Health 15:214-224, 1967.

APPENDIX 14A

ANNOTATED COMMENTS ON COMMUNITY HEALTH EPIDEMIOLOGICAL STUDIES NOT DISCUSSED IN DETAIL IN MAIN TEXT OF CHAPTER 14

APPENDIX 14A

Many community health epidemiological studies have been cited during the past 10 to 20 years as providing quantitative evidence for particular atmospheric levels of sulfur oxides and/or particulate matter being associated with mortality or morbidity effects. In the course of the present assessment, close examination of such studies and published evaluations or reinterpretations of their findings have led to the conclusion that methodological considerations or published results reported for many of them substantially limit or preclude their usefulness in helping to define qualitative or quantitative air pollution-health effects relationships as part of present criteria development purposes focusing mainly on effects of SO_2 or PM at levels below 1000 μg/m^3. Based on this, many studies were excluded from detailed discussion or consideration in the main text of Chapter 14 or mentioned only briefly in support of certain points made in the chapter. Provided below are annotated comments addressing limitations of various studies for qualitatively characterizing the health effects of PM and SO_2 or for quantifying exposure levels at which such effects occur.

A. STUDIES OF MORTALITY EFFECTS OF ACUTE EXPOSURES

1. British, European, and Japanese Studies

Numerous studies concern early (1950s-60s) severe air pollution episodes in England, when atmospheric concentrations of particulate matter (BS) and sulfur dioxide were very markedly elevated. These include studies by Logan (1953), Scott (1953), Ministry of Health (1954), Wilkins (1954 a,b), Gore and Shaddick (1958), Burgess and Shaddick (1959), Clifton et al. (1960), and Scott (1963). These studies are mainly useful in indicating mortality effects occurring at BS or SO_2 levels well in excess of 1000 μg/m^3 and are widely accepted as such, regardless of particular methodological flaws or limitations associated with each.

Biersteker (1966) also reported on a study of a high pollution episode in Rotterdam in December 1962. During the episode, 24-hour mean concentrations were recorded for particulate matter and sulfur dioxide at about 500 μg/m^3 and 1000 μg/m^3, respectively (OECD smoke/sulfur dioxide methods), together with increased hospital admissions for the elderly (over 50 years old) with cardiovascular diseases and weak indications of increased mortality. However, these results were observed only once in Rotterdam and could have been due to other causes. Further, they do not provide evidence of a strongly convincing statistical relationship between observed mortality and hospital admissions and the air pollution levels reported. It is not possible to determine precisely what the reported smoke levels (in μg/m^3) mean in terms of actual particulate matter mass present in Rotterdam at the time (site-specific calibrations for Rotterdam suggest that the actual smoke levels were about twice the level indicated by the OECD standard curve or ≅1000 μg/m^3). Also, the distance of air monitoring site(s) used from study population residences or the admitting hospitals was not specified, making it impossible to estimate how representative the aerometric data were of population exposures.

A study of relationships between mortality and air pollution in Osaka Japan, was reported by Watanabe (1966). Increases in mortality (about 20 percent) appeared to occur when daily concentrations of PM (as measured by a light scattering method) exceeded 1000 μg/m^3 (4-day average) in association with SO_2 (probably sulfation method) levels of 250 μg/m^3. Low temperatures may have contributed to the effects, and it is not possible to assess with confidence the statistical relationship between observed mortality and the reported pollutant levels which were apparently based on data from a single monitoring station. Nor is sufficient information available by which to interpret the meaning or precision of the reported suspended PM measurement results.

2. American Studies

A long series of studies, mainly employing regression analysis techniques, attempted to define relationships between daily mortality and variations in particulate matter and SO_2 in New York City during periods of the 1950s, 1960s, and 1970s (Greenburg et al., 1962; Glasser et al., 1967; Hodgson, 1970; Glasser and Greenburg, 1971; Schimmel and Greenburg, 1972; Buechley et al., 1973; Schimmel and Murawski, 1976).

Among the disadvantages of these studies is the fact that only data from a single air pollution station in Manhattan were used to correlate changes in air pollution with mortality in the city, raising questions regarding how representative those aerometric measurements are of exposures for the study population drawn from the entire New York City area. Goldstein and Landowitz (1975, 1977a,b) found that most correlations between pollutant levels recorded at any two New York City monitoring stations were $\leq$0.40 and concluded that use of aerometric data from any single station generally did not adequately represent pollutant levels for the entire city. A possible exception to this might occur during severe air pollutant episodes when very markedly increased pollutant levels at all stations may tend to approach a common ceiling elevation but no specific evidence supporting this hypothesis has been reported. Lastly, many criticisms have been advanced regarding specifics of the statistical approach employed (e.g., some correlations likely to be significant by chance alone from among a very large number run) and some of the same and other investigators have since reinterpreted these studies as generally not providing evidence of any association between mortality and SO_2 levels (Buechley, 1975; Schimmel, 1978) and only very weak associations with PM levels (Schimmel, 1978). Generally then, little or only weak evidence can be derived from these studies for excess mortality having been associated with either elevated SO_2 or particulate matter levels in New York City, with the possible exception of observations during severe pollution episodes which suggest effects mainly on the elderly and those with chronic cardiovascular or respiratory system diseases.

B. MORBIDITY EFFECTS AND ACUTE EXPOSURES

1. British, European, and Japanese Studies

Carne (1964) reported a study of general practitioners records of illness during the winter of 1962-63 in London which suggests evidence of morbidity effects among elderly patients during the December, 1962, pollution episode when levels of smoke and SO_2 markedly exceeded 1000 μg/m^3. However, the health effects results cannot be reliably linked quantitatively to specific pollutant levels.

In another British study (Angel et al., 1965), acute respiratory illness attack and prevalence rates in a working population of London appeared to correlate with weekly peaks of pollution measured at several nearby sites. No clear conclusions were stated by the authors regarding the pollutant levels associated with notable illness increases. However, based on the reported data only some apparent increases seemed to occur, generally when smoke and SO_2 levels exceeded 1000 μg/m^3.

Emerson (1973) carried out weekly spirometry tests on 18 bronchitic patients in London during 1969-71 and found no correlation of effects with air pollution levels. However, the author noted that pollutant levels were generally too low to expect any notable effects, and the small number of subjects studied would tend to preclude demonstration of statistically significant differences.

Kevany et al. (1975) studied cardio-respiratory hospital admissions in Dublin during 1972-73 and found low but significant correlations between cardiovascular admissions and SO_2 or BS levels in winters. However, the distances of the air monitors used from study population residences or admitting hospitals were not reported, making it impossible to determine the representativeness of the exposure estimates employed.

Derrick (1970) conducted a study of nighttime emergency room visits for asthma in Brisbane, Australia and reported negative correlations between asthma visits with degrees of smoke shade. However, day-of-week effects were not analyzed, and information on methods was insufficient to determine location of air monitors in relation to study population residences or hospitals included in the study.

Kalpazanov et al. (1976) studied daily incidence of influenza in Sofia, Bulgaria during 1972 and 1974-75 and reported the incidence to be significantly correlated with SO_2 and dust levels. However, the pollutant levels were determined from a single station in Sofia; SES status and crowding were not controlled for in the analysis.

Another report (Gervois et al., 1977), in French, gives little methodological information regarding the pertinent aerometric or health measurements used, and it is impossible to estimate where the air monitors were located in relation to residences of the study populations. A positive association was reported between pollution concentration and acute respiratory disease within one town (Quavrechoin) but not another (Denoin), although there were no clear differences in pollutant levels between the towns.

Studies in the Netherlands reported by van der Lende and colleagues (van der Lende et al., 1969, 1972, 1973, 1975) compared lung function in a large population group in 1969 and again in 1972. In a more polluted area, age-, health-, and smoking-adjusted $FEV_{1.0}$ values in men increased from the first to the second survey rather than decreasing with age as expected. This was reported to be associated with a concurrent decrease in air pollution concentrations, although no information was provided on proximity of monitoring sites to study population residences or workplaces. The investigators considered other possible causes of the improved pulmonary function but concluded that the most plausible was the effect of reduced air pollution. However, little hard evidence was advanced to support this hypothesis. In fact, the changes in pulmonary function test (PFT) results observed over time are as likely due to different experimenters' performing the PFT tests from the earlier to later years.

Van der Lende et al. (1981) recently reported further results for the cohort studied longitudinally for more recent years, when pollutant levels continued to decrease in the Dutch communities of Vlaardingen and Vlagtwedde. They demonstrate that after removing the effects of age, smoking and sex, the residents of the more polluted city (Vlaardingen) exhibited a statistically significant greater decline in VC and FEV_1 in the past nine years than did the residents of the clean community (Vlagtwedde). They state that the association between this more rapid decline in lung function capacity with age and exposure to air pollutants can be seen when slopes of individual regressions for VC and FEV_1 against increasing age are based on several PFT measurements repeated over a long time, even when this effect is not seen in cross sectional studies. The authors also noted that the PFT data were gathered by a large team of trained collaborators over a short time period in the same month each time and suggested that effects observed represent adverse health effects of PM and SO_2 levels present during the nine years of the study in Vlaardingen. The annual average arithmetic means and maxima for smoke (OECD method) during those years were reported to range from 19 to 26 $\mu g/m^3$ and 47 to 130 $\mu g/m^3$, respectively; comparable annual average arithmetic means and maxima for SO_2 were 76 to 106 $\mu g/m^3$ and 136 to 564 $\mu g/m^3$, respectively. However, these concentrations were recorded at the Vlaardingen town hall, an unspecified distance from the residences of study cohort members and the smoke levels were based on the OECD standard curve rather than site-specific calibrations in Vlaardingen per se and, therefore, cannot be used to quantitatively estimate effect or no-effect levels for PM.

Kagawa and Toyama (1975) and Kagawa et al. (1976) studied respiratory functions in 20 Tokyo schoolchildren and observed, after controlling for temperature, significant correlations of acute respiratory decrements with various pollutants measured at a monitoring site on the third-story roof of their schoolhouse. However, only 2 of the 20 children appeared to respond to SO_2 changes and 1 of 20 to suspended PM changes. It is not possible to determine the physical meaning of the PM measurements in terms of $\mu g/m^3$ concentrations based on the light scattering methodology information provided.

2. AMERICAN AND CANADIAN STUDIES

Greenburg et al. (1962, 1963, 1964) reported on studies of air pollution-morbidity relationships in New York City. Peak pollution levels during these studies were often much higher than 1000 $\mu g/m^3$, especially when 1-hour values are considered. Also, methods of analyses preclude clear quantitative statements on pollutant/health effect relationships, especially since a single monitoring station was used to generate aerometric data employed in the study analyses for all of New York City.

Glasser et al. (1967) investigated emergency room visits in seven New York City hospitals during the November 1966 air pollution episode. They reported that increased emergency room visits for asthma occurred in three of seven hospitals studied. The authors stated that the results were inconclusive, and only a single monitoring site was used for aerometric data employed in the study to estimate population exposures for the entire New York City area.

Chiaramonte et al. (1970) studied emergency room visits at a Brooklyn hospital during the November 1966 air pollution episode. They reported a statistically significant increase in emergency room visits for respiratory disease symptoms, continuing to 3 days after the peak air pollution concentrations. However, three important SO_2 data points were missing from the analysis.

Reports by McCarroll et al. (1966), Mountain et al. (1968), Cassell et al. (1969) and Thompson et al. (1970) on the Cornell study in New York City gave results for correlations of CoHs and SO_2 levels with numerous health end-point measurements. Many of the effects measured, such as tearing, are of modest health significance. Also, numerous analyses are reported, predominantly multiple regression analysis; and coefficients frequently change sign and magnitude as variables are added and deleted, making it difficult or impossible to quantify health effects/air pollution relationships from most of these studies.

Both Burrows et al. (1968) and Carnow et al. (1969) studied chronic respiratory disease patients in Chicago in relation to SO_2 and PM. Burrows et al. (1968) obtained aerometric data from a Chicago Loop area station and compared this to symptom responses reported by a changing subject pool and concluded that in Chicago "air contaminants do not appear to play a role in producing these symptomatic exacerbations." But he felt guarded about this conclusion considering the several limitations noted in the study. Carnow et al. (1969) developed a personal air pollution exposure index using a network of stations and found that positive associations between SO_2 levels and health effects were restricted to seriously ill elderly patients over age 55 with severe bronchitis. The representativeness of the exposure assessment is uncertain.

Heimann (1970) studied acute morbidity and mortality during air stagnation periods in Boston episodes for 1965 and 1966. Respiratory patient visits to hospitals (but not mortality) were found to increase during the episodes. Even though a monitoring network was used, the representativeness of the exposure data is difficult to determine. TSP levels were below the 24-hour standard.

Lebowitz et al. (1974) conducted a study in an Arizona town to analyze the relationship between air pollutants and health effects observed in exercising elementary students and other groups. The monitoring site for the main study was a quarter of a mile from the school exercising site in Tuscon. Appropriate pulmonary function testing of the subjects was done, and the statistical analyses appeared to be appropriate. However, a pollution index was utilized in characterizing high and low pollution levels that included temperature, relative humidity, TSP and sulfates, and this did not allow for any quantitative attribution of observed effects to specific air pollutants or levels, although TSP levels ranged from 103 to 117 $\mu g/m^3$. Also, only a limited number of data points were obtained and the exercise levels of the subjects were not characterized.

Dodge (1980) reported the results of a cross sectional respiratory health study of over 700 school children living in or near nonurban smelter communities in Arizona. When the subjects were grouped by area of residence, the children living in areas with relatively high particulate levels had significantly lower pulmonary function levels than children living in areas with low particulate levels. The two high TSP level communities had annual averages of 72 and 76 $\mu g/m^3$ with peak 24-hour levels of 119 and 550 $\mu g/m^3$. The low TSP level communities had annual averages ranging from 37 to 54 $\mu g/m^3$ and peak levels ranging from 84 to 125 $\mu g/m^3$. Sulfur dioxide 24-hour peak levels ranged from 700 to 3658 $\mu g/m^3$, with the highest SO_2 levels occurring in the low TSP communities. A standardized questionnaire and pulmonary function testing method were utilized. Ethic and social class analysis were conducted and the effects of parental smoking habits were examined. However, the author noted several concerns that include: families with members that develop respiratory problems tend to leave such industrial communities so that the sample may be from a population with abnormally high pulmonary function; the low TSP areas have high SO_2 levels, so that it might be stated that the high SO_2 areas have significantly higher pulmonary function results; the study results may not generalize to the U.S. population since the smelter and local fugitive dust may make these study areas unique; and longitudinal studies are needed to determine the importance of the findings. Additionally, neither an adequate discussion of aerometric methods nor a discussion of the representativeness of the exposure estimates were provided in this report.

Ramsey (1976) tested pulmonary functions daily in seven male nonsmoking asthmatic (ages 19-21) students at the University of Dayton, Ohio, over three months. Multiple regression analyses showed significant correlations for some tests in 5 subjects with weather variables, but not with TSP or ozone monitored at a site on the University campus. However, no details are provided regarding the air pollution measurement methods and the small number of students studied tends to preclude finding of statistically significant effects unless they were rather marked ones. Nor was information provided on any control for possible medication effects on the study results.

Whittemore and Korn (1980) reported on newly developed statistical methods, the use of which were illustrated by use of data collected as part of the EPA CHESS program concerning panels of asthmatics living in the Los Angeles area. The analysis examines daily asthma attacks among 443 asthmatics during 34 week periods during the years 1972-1975. Data for 24-hour average concentrations of TSP, respirable suspended particulates, suspended sulfates and nitrates, SO_2, NO_2 and photochemical oxidants were utilized. Neither the quality nor the representativeness of this aerometric data is certain. A separate multiple logistic regression was used for each panelist's asthma attack data. The presence of an attack on the preceding day was the most significant predictor of attack. Most attacks occurred on days with high oxidant and particulate pollution, cool days, and on Saturdays (the last day of the weekly reporting period). Because TSP, RSP, NO_x and SO_x were highly correlated, TSP was used as a surrogate for this group of pollutants. The analysis did not use data on medication use nor pollen counts. The most serious limitation is the absence of information on individual pollutant exposure. Problems (including an inadequate exposure estimate, use of TSP as a surrogate for pollutants, and high attack rate on last reporting day of the reporting period) limit the usefulness of the study for present purposes; however, the statistical analysis approach utilized (looking across time within an individual) is an important development that may prove beneficial in future studies.

Measurements of air pollution levels for an air pollution episode in Pittsburgh, Pa., reported on by Stebbings et al. (1976, 1979), were compared to health endpoints measured (mainly pulmonary function, e.g. FEV tests) after the highest levels of pollution had occurred. Method of subject selection, lack of clear association of health results to specific particulate matter (TSP) or SO_2 levels, and lack of baseline health data preclude quantitative conclusions regarding health effects-air pollution relationships.

Becker et al. (1968) studied the relationship of air pollution changes to responses of 2,052 white collar workers on a questionnaire administered within 48 hours of the November 23-27, 1966 episode in New York. SO_2, CO, and smoke shade (CoHs) were measured at various locations with a peak for SO_2 of 0.55 ppm. How representative the aerometric measurements are for the group studied is uncertain, but the greatest number of positive responses for six of eight respiratory symptoms occurred on peak episode days.

Dockery et al. (1981) presented preliminary results of a study of pulmonary function changes in children associated with acute exposure to TSP and SO_2 during air pollution episodes in Steubenville, Ohio. Initial results of four (when 24-hour TSP levels rose from 150-200 to 400 $\mu g/m^3$) study periods are suggestive of some decrements in pulmonary function occurring in children during episodes and persisting for some time (3-4 weeks) thereafter. However, the results thus far are somewhat ambiguous and both the study and analyses are continuing as part of the previously described "Six-City Study," which appears to have reasonably representative exposure estimates; unfortunately, this preliminary brief report does not allow for clear conclusions to be drawn yet.

A Canadian study (Levy et al., 1977) related hospital admissions records for acute respiratory conditions to changes in average weekly levels of an air pollution index (combining CoH and SO_2 measurements from a single monitoring site in Hamilton) and found significant correlations after effects of temperature were taken into account. However, no clear quantitative associations between specific CoH or SO_2 levels and increased acute respiratory conditions could be deduced from the reported results.

C. MORTALITY AND CHRONIC EXPOSURES

1. British, European, and Japanese Studies

Pemberton and Goldberg (1954) studied 1950-1952 bronchitis mortality rates in men 45 years of age and older in county boroughs of England and Wales. They reported that sulfur oxide concentrations (sulfation rates) were consistently correlated with bronchitis death rates in the 35 county boroughs analyzed. However, smoking and occupational exposures were not controlled for in the analysis.

Gorham (1958, 1959) studied 1950-1954 deaths in 53 counties of England, Scotland, and Wales and reported that bronchitis mortality was strongly correlated with acidity of winter precipitation. However, no correlations with more direct measurements of SO_2 or PM were provided; nor is it possible to evaluate the quality or precision of acidic precipitation measurements used in the study.

Studies by Stocks (1958; 1959; and 1960a,b) investigated associations in the 1950s between standardized mortality ratios for bronchitis, lung cancer, and other cancers and particulate matter (BS) levels in 101 urban and rural areas of Wales and England, but they provide no way to determine clearly quantitative relationships between BS and mortality effects. Interpretation of the BS aerometric data alone is clouded by ambiguities regarding the actual mass of BS (in $\mu g/m^3$) indexed by measurements reported for various areas of England and Wales (i.e. were site-specific calibrations used to make the $\mu g/m^3$ estimates?). Also, perhaps more importantly, differences in smoking history were not assessed as possibly accounting for reported urban-rural differences.

Burn and Pemberton (1963) studied 1950-1959 deaths from all causes, bronchitis, and lung cancer in three polluted areas of Salford, United Kingdom. The gradient of mortality from all causes and from bronchitis and lung cancer followed the pollution gradient. However, no direct correlation was provided specifically with measurements of SO_2 or PM in the study areas, nor was smoking or SES taken into account in the analysis of study results.

Also, Buck and Brown (1964) attempted to relate standardized mortality ratios from 214 areas in the United Kingdom (1955-59) to daily smoke (BS) and SO_2 levels for March, 1962. Several factors make it difficult to interpret or accept the results of this study including: (1) pollution levels for 1962 do not provide an adequate basis for quantitatively estimating what were probably much higher BS and SO_2 levels possibly contributing to mortality occurring in 1955-59; (2) it is not clear what the reported 1962 BS data in $\mu g/m^3$ mean in terms of actual mass indexed from the various U.K. areas, including most for which no site-specific calibrations were carried out; (3) the 1962 BS levels calculated were likely affected to an unknown extent by a computer error reported by Warren Spring Laboratory for British National Survey BS data during 1961-64; and (4) potential effects of differences in occupational exposure were not taken into account.

In another study (Wicken and Buck, 1964) of cancer and bronchitis in 12 areas of England, no actual measurements of particulate matter (BS) or SO_2 were available except for two areas

(North and South Easton) during 1963-1964 and an effort was made to relate these levels to mortality during 1952-1962. Similar objections to retrospective linking of later aerometric data to earlier mortality information apply here as were stated above for the Buck and Brown (1964) study. Also, mortality differences for the two Easton areas may have been due as easily to differences in study population age levels and smoking patterns as to any air pollution gradient.

Lave and Seskin (1970) attempted to demonstrate, by mathematical analyses mainly involving regression analyses, relationships between BS measurements in the United Kingdom and bronchitis mortality data once the effects of other factors, such as smoking and socioeconomic status (SES) are removed. This Lave and Seskin (1970) study has been extensively critiqued in detail by others (Holland et al., 1979) who have noted, for example, difficulties in justifying inclusion of smoking, SES, and air pollution levels in the Lave and Seskin analyses as if they were completely independent variables and the failure to make some direct allowance for smoking habits in the actual analyses. Perhaps even more basic difficulties with the analyses derive from: (1) the use of qualitative BS aerometric data not appropriately translated into mass concentration estimates (in $\mu g/m^3$) for smoke by means of site-specific calibrations of reflectance readings against local gravimetric measurement data; and (2) ambiguities regarding the location of sampling devices in relation to study population residences or work places and, therefore, the representativeness of the aerometric data in estimating population PM exposures.

Collins et al. (1971) studied death rates in children 0-14 years of ages, 1958-1964, in relation to social and air pollution indices in 83 county boroughs of England and Wales. Partial correlation analysis suggested that indices of domestic and industrial pollution account for differences in mortality from bronchopneumonia and all respiratory diseases among children 0-1 year of age. However, a visibility index was used as the basis for development of the air pollution index and not BS or SO_2 measurement data *per se*, making it impossible to link the observed mortality findings quantitatively to those air pollutants.

Kevany et al. (1975) studied 1970-1973 deaths from various causes in Dublin, Ireland. Partial correlation analysis yielded significant associations between air pollutants and some specific causes of death. Although a multistation air sampling network was reportedly used to derive exposure estimates, it is unclear as to whether site-specific calibrations were employed for BS data or how representative the monitoring sites were in relation to study population residences or workplaces.

Lindeberg (1968) studied deaths during Oslo winters in relation to air pollution indices. Average deaths per week, in the 1958-1965 winters, were reported to be correlated with air pollution. However, the reported information neither allows for judgments to be made regarding the representativeness of the aerometric data in relation to estimating study population exposures, nor adequately takes into account certain other factors, such as socioeconomic status, in the data analysis.

Toyama (1964) studied mortality rates in districts of Tokyo and reported that bronchitis mortality (but not cardiovascular, pneumonia or cancer mortality) was associated with dustfall levels. Insufficient information was provided by which to judge how representative even the crude dustfall measurements of PM levels were in relation to study population residences or workplaces.

Watanabe and Kaneko (1971) studied 1965-1966 mortality by cause in three areas of Osaka, Japan. They reported a stepwise increase in mortality to be related to air pollution independent of temperature. However, the PM measurement data were simply reported to have been obtained for study areas from Institute of Hygiene records, and insufficient information was provided by which to estimate the representativeness of the data in relation to study population residences or workplaces.

2. American and Canadian Studies

Among published American reports on mortality associations with chronic PM and sulfur oxides exposures are several regarding results from the Nashville, Tennessee, air pollution study (Zeidberg et al., 1961; 1967; Hagström et al., 1967). One purpose of the Nashville study was to study relationships between air pollution levels and mortality (total and respiratory disease-specific) in areas of the Nashville, SMSA. Particulate matter and sulfur oxides measurements obtained during 1958-1959 were related to deaths occurring during 1949-1960, opening this study to criticisms regarding retrospective use of later aerometric data to look for links with earlier mortality. Also, data regarding smoking habits and occupational exposures were not taken into account in these studies.

Lepper et al. (1969) studied 1964-1965 mortality rates in Chicago census tracts stratified by socioeconomic class and SO_2 concentration. Increased respiratory disease rates were seen in areas of intermediate and high SO_2 concentration, within a socioeconomic status, without a consistent mortality gradient between the areas of intermediate and high SO_2 concentration.

Another set of American mortality studies was conducted in Buffalo, N.Y., by Winkelstein and associates (Winkelstein and Kantor, 1967; Winkelstein et al., 1967, 1968; Winkelstein and Gay, 1971). Numerous criticisms of these studies bave been discussed by Holland et al. (1979) and Ware et al. (1981). Among the more salient problems noted were: (1) the use of 1961-1963 particulate matter and sulfur oxides measurement data in trying to retrospectively relate air pollution to mortality among the elderly during 1959-1961; (2) inadequate controls for possible age differences between study groups that may have covaried with the air pollution gradient used; (3) lack of information on lifetime, including occupational, exposures; and (4) failure to correct for smoking habits. In a later presentation, Winkelstein (1972) commented on several of these points and attempted to correct for some of them, such as by looking at smoking patterns among certain populations living in some of the same study areas included in the earlier analyses. Still, this 1972 discussion does not lay to rest many of the different major concerns regarding mortality findings reported by Winkelstein for Buffalo. For example, the finding of no significant differences in smoking patterns among study areas in the follow-up investigation (Winkelstein, 1972) does not adequately control for possible smoking effects among differing population cohorts included in the earlier study analyses reported by Winkelstein and coworkers.

Jacobs and Landoc (1972) studied 1968-1970 mortality rates in industrial versus nonindustrial areas of Charleston, S.C. Higher total and heart disease mortality rates were found in industrial areas, but socioeconomic differences were not adequately controlled for in the analyses. Nor were TSP measurements used likely representative of study population exposures.

Morris et al. (1976) studied 1960-1972 mortality rates compared to 1959-1960 air pollution levels in Seward and New Florence, Pa. They reported that mortality could be related for both smokers and nonsmokers to air pollution exposures. However, socioeconomic status and occupational exposures were not adequately controlled for, despite a much larger percentage of the study population being coal miners or steelworkers in New Florence, the more polluted community.

Two further publications by Lave and Seskin (1972, 1977) attempted to extend their original U.K. analysis approach (Lave and Seskin, 1970) to metropolitan statistical areas in the United States. Many similar comments as indicated above for the earlier Lave and Seskin (1970) publication apply here. Of crucial importance is the basic difficulty encountered in trying to determine how representative the air pollution data used in the analyses were of the actual exposures of individuals included in their study populations. For example, in some instances data from a single monitoring station were apparently used to estimate pollutant exposures for study populations from surrounding large metropolitan areas. No clear information on quantitative relationships between particulate matter or sulfur oxides levels and mortality can be derived from these published analyses. Similar criticisms and others noted earlier in the main text apply to the recent Chappie and Lave (1981) report.

Thibodeau et al. (1980) do not disagree with the conclusions of Lave and Seskin regarding mortality being associated with chronic exposure to PM, SO_2, or sulfates; however, Thibodeau et al. (1980) express reservations about the methods of estimating the magnitude of the association and see the Lave and Seskin work as a prioritizing step leading to further research that will more directly examine a possible cause-and-effect relationship. The Thibodeau et al. (1980) reanalysis of U.S. mortality data demonstrated that the regression coefficients for air pollution variables were quite unstable and should be used with care when interpreting their meanings. Christainsen and Degen (1980) also examined the work of Lave and Seskin and felt that the hypothesis of a constant relationship between levels of air pollution and the explanatory variables should be rejected, which they feel weakens the validity of the Lave and Seskin results; however, they state that their results still support the contention that air pollution bears a significant relationship to mortality rates.

Lipfert (1980) presented three mortality analyses for: (1) air pollution episodes; (2) time-series analysis; and (3) cross sectional analysis over a large number of U.S. cities.

Based on those analyses, Lipfert concluded that no reliable statistical association existed between air pollution and excessive mortality. However, the method and representativeness of the exposure estimates are not clear and occupational exposure is not examined. Mendelsohn and Orcutt (1979) similarly examined a large death certificate data base to explore the relationship of pollutants and mortality. However, they compared 1974 air quality data to 1970 mortality data and did not examine smoking habits, significantly weakening the meaning of any results reported by them.

Schwin and McDonald (1976) also studied a pollutant-mortality data base and, utilizing ridge regression and sign constrained least squares analysis, concluded that increased concentrations of sulfur compounds are associated with a general increase in the total white population mortality rate. A total of 23 independent variables were used, including: socio-economic status, climate, pollution and cigarette smoking. Cigarette smoking was indexed by state per capita sales, a very crude proxy for actual smoking data for individuals. The method and representativeness of the exposure estimates are also not clear, lessening confidence in the reported findings.

Mazumdar and Sussman (1981) studied mortality and morbidity in relation to air pollutants in Pittsburgh, Pennsylvania, and reported significant associations between PM (CoHs) but not SO_2 and mortality in Allegheny County. A time-series analysis of abstracted hospital discharge records for the time period 1972-1977 was also carried out to assess morbidity effects, with significant associations being found between both PM and SO_2 and increased rates of respiratory, heart, and other circulatory diseases. Three monitoring stations were used to yield exposure estimates, a clear improvement over the fewer sites used in many other studies. Also, temperature was controlled for and a 15-day moving average was used to remove cyclic components of periods greater than fifteen days, making for a generally credible study overall. However, the rough terrain in Pittsburgh and the large geographical size of Allegheny County raise concern over the representativeness of the exposure estimates based on data from three monitoring sites. The authors also note questions concerning the relationship between data from discharge records and the patients' hospital records which might affect the classification according to disease type and indicate that further verification of disease classification is being carried out now. Once accomplished, this study may provide some useful information for criteria development purposes.

D. MORBIDITY AND CHRONIC EXPOSURES

1. British, European and Japanese Studies

Several British studies are often cited as demonstrating morbidity effects to be associated with chronic exposures of particulate matter (BS) and sulfur oxides. For example, Fairbairn and Reid (1958) carried out comparisons of respiratory illness among British postmen living in areas of heavy and light pollution. Sick leave, premature retirement, and death due to bronchitis or pneumonia were closely related to a pollution index based on visibility. The morbidity findings, however, were not analyzed in relation to more direct measures of SO_2 or PM; thus, no means were provided to relate pollution indices data to population exposures to PM or SO_2.

Mork (1962) employed questionnaire and ventilatory function tests in studying male transport workers 40-59 years of age in the Norwegian city of Bergen, and London, England. Greater frequency of symptoms and lower peak flowrates were found in London. These differences were not explained by smoking habits or socioeconomic factors and were therefore hypothesized to result from differences in air pollution levels. However, no direct associations were demonstrated between morbidity effects and monitored BS and SO_x levels. Nor could one determine the representativeness of any aerometric data used in relation to the study populations.

Another study by Burn and Pemberton (1963) demonstrated increases in sickness absences for bronchitis in association with increases in BS levels. However, this study failed to test for effects of temperature decreases, which covaried with the occurrence of pollution increases and did not adequately control for smoking effects.

In another study (Ministry of Pensions and National Insurance, 1965), rates of illness, and absences for diseases such as bronchitis were related to smoke (BS) and SO_2 measurements from six areas of England, Scotland and Wales in 1961-1962, yielding apparent correlations between bronchitis and pollutant levels in some areas but not in others. However, socioeconomic and several other possible confounding factors were not taken into account. Also, BS aerometric measurements were likely influenced by a computer error affecting 1961-1964 British National Survey BS data, and one cannot determine what the reported BS levels actually mean in terms of $\mu g/m^3$ mass estimates in the absence of information on site-specific mass calibrations for the widespread localities studied.

Douglas and Waller (1966) and Kiernan et al. (1976) reported on a long-term prospective longitudinal study of the association of respiratory disease and symptoms in a cohort of children in the United Kingdom in relation to air pollution. This widely cited prospective study examined a group of over 3,000 children with adequate health effect endpoint measurements and appropriate statistical analysis. However, no direct measurements of concurrent air pollutant levels were used; rather only a very crude estimate of likely smoke or SO_2 levels based on a coal consumption index was employed and retrospectively partially confirmed by later aerometric data in some study areas. The crudeness of the exposure assessment does not allow for quantitative determination of the levels of either PM or SO_2 (or any other pollutant) possibly associated with the reported health effects, and even the classification of study areas from "low" to "high" pollution areas can be seriously questioned in the abscence of any direct confirmatory data reported for the years studied.

Holland et al. (1969a,b) and Bennet et al. (1971) studied the prevalence of respiratory symptoms, ventilatory function, and past histories of respiratory illness in more than 10,000 schoolchildren 5-16 years of age residing in four different areas of northwest London (Kent), 1964-1965. Childhood smoking habits and degree of air pollution were found to have the greatest influence on respiratory symptom prevalence and ventilatory function. Social class, family size, and past history of respiratory disease were also found to contribute to the observed health effect. All factors operated independently and exerted their effects collectively. However, aerometric data obtained 1-3 years after the health data were employed retrospectively to estimate study population exposures, precluding quantitative conclusions regarding morbidity/air pollution relationships.

Another study by Holland et al. (1969c) examined respiratory disease and pulmonary function in families of Harrow, England (a suburb of London) during 1962-1965. During this period, mean winter smoke levels declined in $\mu g/m^3$ from 108 (1962-1963) to 72 (1964-1965) in two "clean" areas and from 175 (1962-1963) to 73 (1964-1965) in two "dirty" areas, but SO_2 levels for the same areas, respectively, were higher in 1963-1964 and 1964-1965 for the "dirty" areas than the "clean" ones. The study was generally well conducted and controlled for many potentially confounding factors. However, the observed differences in respiratory symptoms may have been quantitatively related to earlier higher pollutant levels and the lack of site-specific BS calibrations precludes any quantitative conclusions regarding effective BS levels expressed in terms of $\mu g/m^3$ mass concentrations.

Colley and Reid (1970) also studied respiratory disease prevalence in more than 10,000 children 6-10 years old in England and Wales during 1966. A definite gradient for past bronchitis and current cough was found from lowest rates in rural areas to highest rates in the most heavily polluted urban areas. Differences were clearer in children of semiskilled and unskilled workers. No effect on upper respiratory illness rates was seen. However, the air pollution levels were qualitatively estimated from aerometric data that is not adequately described. Other problems are also posed regarding exposure estimates, e.g., two of five rural areas used had no air monitoring stations.

Lambert and Reid (1970) analyzed data from a respiratory symptom questionnaire mailed to about 10,000 Britains in relation to pollutant levels of SO_2 and smoke. Analysis of the health data was appropriate and considered major confounding factors. However, study areas for only 30 percent of the sample had smoke (BS) and SO_2 data derived from the National Air Pollution Survey conducted in 1975, and the rest were based upon estimates from the Douglas and Waller coal consumption index. In addition, the study did not adequately assess possible confounding effects due to occupational exposures.

Gregory (1970) studied sickness absenteeism for Sheffield, United Kingdom steelworkers in the 1950's and reported correlation of weekly absences with SO_x and PM levels. However, smoking was not controlled for, and the author states that the number of aerometric data points available during the winter studied were too few to allow firm conclusions to be drawn.

Waller et al. (1974) studied ventilatory function and respiratory symptom prevalence among 18-year-olds born in London just before and after the smog episode of 1952. No differences were found between the two groups. Both were exposed later to a high degree of pollution during the 1950s. History of lower respiratory illness in childhood did have a major influence on later symptoms and ventilatory function, but effects were not attributable clearly to either SO_2 or BS.

A study by Fletcher et al. (1976) concerned relationships between smoke (BS) levels and morbidity effects in working men in London. However, although certain apparent relationships were detected, the authors (Fletcher et al., 1976) noted several factors which complicate interpretation of their findings in such terms, and it is difficult to link observed effects to quantitative levels of BS.

Preliminary reports from a study not yet completed of children in many areas of the United Kingdom have been presented by Irwig et al. (1975). The results reported, however, are not based on final analyses of data and have not been subjected to peer review. Also, aerometric data for BS measurements employed appear to be based on mass estimates derived from calibration of reflectance readings for the British National Survey standard curve (for London air in 1963) not necessarily accurately reflecting actual BS mass levels in $\mu g/m^3$ existing in non-London study areas included in the analyses. The study, then, cannot be expected to yield any meaningful information even when completed .

Biersteker and van Leeuwen (1970a,b) studied respiratory symptoms and (pulmonary) functions in schoolchildren in relation to pollutant levels in Rotterdam. The data from approximately 1,000 children were analyzed by appropriate statistical methods. However, insufficient details on the aerometric measurements were presented to determine the representativeness of the exposure assessment, and socioeconomic status was not studied.

A series of studies from Poland by Sawicki (1969a,b; 1972) and (Sawicki and Lawrence, 1977) reported higher prevalence rates of chronic bronchitis in males (all smoking categories) and females (smokers and nonsmokers but not exsmokers) in a high-pollution community. However, many of the reported differences by air pollution gradient disappeared when rates were adjusted for age, sex, and smoking habits. Also, no consistent relationship was found between the chronic bronchitis prevalence rate and length of residence in the high-pollution community. Several reviewers (e.g., Holland et al., 1979) have taken this as evidence indicating that Sawicki's findings do not show a relationship between air pollution and bronchitis. In a repetition of this study in 1973, Sawicki and Lawrence (1977) found some further evidence suggesting a relationship between the prevalence of chronic bronchitis and air pollution levels. By 1973, annual smoke concentrations in the high-pollution area were reported to average 190 $\mu g/m^3$ (OECD) compared with 86 $\mu g/m^3$ (OECD) for the low-pollution area. However, these are essentially meaningless estimates of PM mass concentrations present, given that the OECD standard curve was used rather than local site-specific calibrations in generating the smoke estimates. More accurate estimates of PM mass based on the latter could differ substantially from those derived from the OECD curve. Sulfur dioxide average annual concentrations were 114 and 46 $\mu g/m^3$, respectively, for the high and low pollution areas. Both chronic bronchitis and asthma were more prevalent in the high pollution area in males and females aged 31 to 50 and in smokers. Chronic bronchitis was also more prevalent in female nonsmokers in the high pollution area in both 1968 and 1973. The investigators demonstrated an interaction between air pollution and smoking. However, the authors concluded that air pollution, in comparison to other factors (such as smoking) exerted a relatively minor effect on the health of their study populations. Chanska (1980) discusses additional data from the Sawicki study series, including methods used in the studies to take socioeconomic status into account.

Rudnick (1978), as part of a generally well-designed and methodologically-sound study, collected data by a self-administered questionnaire on respiratory symptoms and disease, as well as by PEFR tests, in 3805 children 8 to 10 years old living in three communities in Poland with differing air pollution concentrations. The questionnaire sought information on respiratory symptoms and asthma symptoms during the previous 12 months. Mean pollutant concentrations in the higher pollution area for the years 1974 and 1975 were 108 to 148 μg/m^3 for SO_2 (OECD) and 150 to 227 μg/m^3 for smoke (OECD). The low-pollution areas had SO_2 concentrations of 42 to 67 μg/m^3 and smoke concentrations of 53 to 82 μg/m^3. However, the smoke readings in μg/m^3 were not obtained by site-specific calibration and, therefore, cannot be used for more than a basis for very rough qualitative comparisons of results from different geographic areas investigated in the study. Most symptoms of respiratory illness in both boys and girls occurred more frequently in the high-pollution area but the differences were, in general, nonsignificant. Adjusted PEFR values were lowest (non-significantly) in children living in Cracow, the high-pollution area, but it is unclear as to how variation due to instrumentation drift with the Wright Peak Flow meters used was minimized. There was a higher prevalence of breathlessness, sinusitis, and asthma attacks in boys living in the high-pollution area, but only "runny nose in the last 12 months" occurred more frequently in girls in the same area. There were no significant differences between the frequencies of nonchronic cough, attacks of breathlessness, shortness of breath, or multiple cases of pneumonia associated with the different pollution levels. These results, however, again cannot be taken as indicative of no-effect levels at the above SO_2 and smoke concentrations, due to ambiguities concerning the measurement methods used (e.g., the lack of site-specific calibrations of the smoke levels) and given that the author concluded that "the results of the main cross-sectional study pointed to dominating effects of living in urban highly polluted environment on the frequency of respiratory symptoms and past respiratory illnesses."

Colley and Brasser (1980) reviewed the results from eight European studies (of which the Rudnik study is one) conducted under the guidance of a WHO working group. Standardization of design and aerometric measurement was sought but not totally attained. The results were not consistent, relationships between air pollution and respiratory diseases or pulmonary function declines being found in some countries but not in others. Nevertheless, Colley and Brasser did data analyses for results from all of the studies pooled together, despite noting numerous caveats and concerns about the appropriateness of such pooled analyses, and concluded that "this study has clearly shown a strong association between air pollution and various respiratory indices in children." They stressed that because of the small number of observations, varying pollutant levels, different prevalence levels in different countries, and problems in pooling the data that "caution must be exercised in drawing any firm conclusions on the precise relationship between levels of air pollution and prevalence of respiratory indices." However, it is questionable whether any meaningful comparisons can be made at all between the different study results in relation to PM effects due to smoke levels in μg/m^3 being determined based on the OECD standard curve rather than site-specific calibrations of gravimetric mass measurements versus reflectance readings in each of the communities studied.

During the winter season of 1977-78, Sâric et al. (1981) studied pulmonary function and respiratory disease in two groups each of more than 70 second grade students, one group living in an industrialized community with elevated air pollution and the other in a cleaner-air community in Yugoslavia. The authors concluded that adverse effects on pulmonary function and respiratory symptoms were associated with exposure to annual average SO_2, smoke, and suspended particulate matter (SPM) levels of 70-80 μg/m^3, 60-80 μg/m^3 and 130-200 μg/m^3, respectively, with frequent exposure to high peak episodes during the heating season. Pollutants were measured both indoor and outdoors at the schools, and activity patterns were considered in evaluating the exposure. However, the smoke data reported were not based on site-specific calibrations and the SPM measurement may not be directly related to TSP measurements, although the SPM measurements were reported to have been obtained by hi-volume samplers. Additionally, not all major pollutants (e.g., NO_2) were measured. Pulmonary function tests were administered each week by the same physician. Symptoms reported on provided postcards were followed by a visit by a nurse to collect details of the respiratory disease. SES status was evaluated. The confounding factors of smoking parents and number of family members were found not to affect the results. Also, no epidemic of any respiratory disease was observed during the study period, reducing the possibility of this confounding the study. In addition, although the respiratory function measurements were taken by the same experimenters in both towns, "blind" procedures were not utilized that would have enhanced confidence in the reported findings concerning the pulmonary function tests. However, the reported decrements in pulmonary function tests are consistent with the other findings of increased incidences in respiratory disease symptoms, the reporting of which was not likely affected by the particular survey procedures employed. This cross sectional study does not allow us to separate the possible long-term effects of exposure to pollutants from exposure around the day of measurement. Besides the limitations mentioned above, the paper does not provide adequate information to consider it in other than qualitative terms since one cannot determine the adequacy of their analysis method, nor the representativeness of their exposure estimates.

Petrilli et al. (1966) studied chronic respiratory illness, rhinitis, influenza, and bronchopneumonia in several areas of Genoa, Italy, in relation to air pollution concentrations, between 1954 and 1964. Respiratory illness rates in non-smoking women over age 64 with a long residential history and no industrial exposure history were strongly correlated with SO_2 concentrations. These investigators found that all illness rates were higher in industrial districts with higher annual mean pollution concentrations. However, differences in socioeconomic status between study areas were not adequately controlled for, and ambiguities exist regarding methodology and interpretation of reported aerometric measurements.

Watanabe (1965) studied peak flow rates in Japanese school children residing in Osaka and found lower peak flow rates in children from more polluted communities. Improved peak flow rates occurred when air pollution levels decreased. However, it is not clear how Watanabe divided study areas into three levels of varying pollution. Nor was socioeconomic status (SES) controlled for in the study.

Toyama et al. (1966) evaluated respiratory symptoms and spirometry in an agricultural area of Japan during 1965 in subjects ages 40-65, categorized by smoking and sex. Much lower prevalence rates and higher lung function levels were found than elsewhere in Japan based on other study results. No direct associations were demonstrated between SO_2 or PM measures and the effects observed; nor were SES factors taken into account.

Tsunetoshi (1968) conducted a bronchitis survey of seven areas of Osaka, Japan, in 1966 among 36,000 adults 40 years of age and over. Bronchitis rates, standardized for sex, age, and smoking were greater among men and women in the more polluted areas and appeared to follow the air pollution gradient. However, it is not clear how the air pollution gradient was determined or how representative the aerometric data used were of study population exposures.

Fujita et al. (1969) conducted a prevalence survey (Medical Research Council questionnaire) of post office employees in Tokyo and adjacent areas in 1962 and resurveyed the same areas in 1967. A two-fold increase over time in prevalence of cough and sputum production was seen in the same persons, irrespective of smoking habits. This change was attributed to increasing degrees of air pollution, but no direct evidence for this hypothesized relationship was provided. Dustfall estimates for 1964 and 1967 alone were used to estimate population exposures, but how representative these data were of study population exposures was not clear based on the data provided.

Yoshii et al. (1969) investigated chronic pharyngitis and histopathological changes in 6th grade children in three areas of Yokkaichi, Japan. Correlation of both effects with sulfation rates were seen. Although 18 monitoring sites were used to obtain the sulfation rate data, the relationship of monitoring sites to population residences is not clear. Also, of 5331 clinic visits only 287 cases were followed up and, of these, 71 were thought to be due to air pollution because no other apparent etiology was identified.

Nobuhiro et al. (1970) studied chronic respiratory symptoms via a survey of high- and low-exposure areas of Osaka and Ako City, Japan. Higher prevalences of chronic respiratory symptoms were seen in more polluted areas based on SO_2 measured by the PbO_3 method and dustfall measured by deposit gauge at 4-6 and 6-19 sites, respectively. However, little continuous data were available during the study periods and SES differences were not controlled for in the study.

Tsunetoshi et al. (1971) conducted a prevalence survey of chronic bronchitis in nine areas of Osaka and Hyogo Prefecture, Japan, in residents aged 40 or older. Multiple regression analysis indicated increasing bronchitis prevalence, adjusted for age, sex and smoking, corresponding to the area gradient of air pollution based on dustfall and SO_2 measurements obtained over a 3-year period prior to obtaining the health data. Insufficient information was provided to allow for estimation of the representativeness of the exposure data in relation to study population exposures.

Yoshida et al. (1976) studied the prevalence of asthma in schoolchildren in areas of Japan. Increased prevalence rates were found in areas with higher sulfation rates, but it was unclear how well the aerometric data related to population exposures.

Suzuki et al. (1978) conducted a prevalence survey of respiratory symptoms in housewives ages 30 or over in Japan. They found that prevalence rates correlated with various pollutants, especially in older smokers. However, the study did not control for SES factors and the authors themselves expressed concern over their poor exposure estimates based on limited, qualitative aerometric data.

Biersteker and van Leeuwen (1970a,b) evaluated peak flow rates and respiratory symptoms in 935 schoolchildren living in two districts of Rotterdam, one relatively affluent and having good air quality (40 μg of smoke per m^3 and 120 μg of SO_2 per m^3) and the other less affluent and having 50 percent higher concentrations of smoke and SO_2. No significant area differences were found in peak flow, adjusted for height and weight. Significantly more childhood bronchitis occurred in the more polluted district, but differences were judged to be due to poor living conditions, because the low-pollution area consisted of higher class residences. Details of the pollutant measurements were not provided; nor could one determine how representative they were of population exposures.

Reichel (1970) and Ulmer et al. (1970) studied respiratory morbidity prevalence via surveys of random samples of populations in three areas of West Germany with different degrees of air pollution. No differences in respiratory morbidity (standardized for age, sex, smoking habits, and social conditions) were found between populations living in the different areas. Unfortunately, no monitoring data were presented and the monitoring methods used to classify pollution areas were unspecified; therefore, no credence can be placed in the results as presently reported by Reichel (1970) or Ulmer et al. (1970).

Zaplatel et al. (1973) studied pulmonary function in 19 schoolchildren in Most, Czechoslovakia and reported that some children living in areas of high air pollution had functional abnormalities. However, no details were given on aerometric monitoring methods or information by which to judge how well such data represented study population exposures to SO_2 or PM.

Ramaciotti et al. (1977) studied bronchitis symptoms and peak flow rates in 1182 men in Geneva in relation to SO_2, smoking, and age in 1972-1976. Regression analyses showed an independent effect of SO_2 after controlling for smoking and age. It is not clear how the air pollution exposure indices were derived or how representative the data were of study population exposures.

2. American and Canadian Studies

Numerous American and Canadian studies are often cited as showing associations between morbidity and chronic exposures to particulate matter or sulfur oxides. Comments on various American studies are as follows.

Dohan and Taylor (1960), Dohan (1961), and Ipsen et al. (1969) studied industrial absenteeism rates and dispensary visits in women RCA workers in several Philadelphia localities and other cities in 1957-1963. Correlations with sulfation rates were explained by temperature and season. However, the authors employed annual mean estimates of an air pollution index developed for an entire city based on TSP, CoHs and suspended sulfates data from an unspecified number of monitoring sites not necessarily in close proximity to either workers' residences or work places. SES was not controlled for in the analysis but may not have been crucial given the use of females generally engaged in similar work capacities for RCA.

Zeidberg et al. (1961) conducted a 1-year study of 49 adults and 34 children with asthma in Nashville, Tennessee, and reported a doubling of asthma attack rates for persons living in more SO_2-polluted neighborhoods. Monitoring data for SO_2 and TSP were obtained from monitoring sites located within one-half mile of study population residences. However, no adjustments for demographic or social factors were made.

Cowan et al. (1963) and Smith and Paulis (1971) evaluated history of asthma and allergen skin tests for University of Minnesota students in relation to exposure to dust from a nearby grain elevator. A significant association was seen between asthma attacks and grain-dust PM exposure as measured by "smoke spot" monitoring on campus, which sampled PM smaller than 5 µm MMAD. This observed association remained significant after effects of temperature, humidity and other possible confounding factors were taken into account in the regression analysis. The study did not allow, however, for clear delineation of effective PM exposure levels inducing increased asthma attacks in the presence of very low levels of SO_2 (40 $\mu g/m^3$).

Prindle et al. (1963) carried out comparisons of respiratory disease and lung function in residents of Seward and New Florence, Pennsylvania, and found increased airway resistance in inhabitants of the more polluted community (New Florence). However, differences in occupation, smoking, and socioeconomic level could account for these observed effects. For example, many more of the residents of New Florence were employed as coal miners or steelworkers, but occupation was not controlled for in the study.

Deane et al. (1965) and Holland and Reid (1965) conducted a questionnaire and ventilatory function survey of outdoor telephone workers 40-59 years of age on the west coast of the United States. No differences in symptom prevalence between San Francisco and Los Angeles workers were found, although particulate concentrations were approximately twice as high in Los Angeles. Also increased prevalence of respiratory symptoms, adjusted for smoking and age, a larger volume of morning sputum, and a lower average ventilatory function were found in London workers compared with American workers. However, the aerometric data used in the studies were simply reported for American cities unspecified as to number or location of monitoring sites; and no information was provided regarding place of residence or work for the study population in relation to the monitors used to estimate PM exposures. In fact, some study subjects may have lived or worked outside the limits of the city for which the monitoring data was reported.

Kenline (1966) analyzed daily visits to the emergency clinic of Charity Hospital, New Orleans, for emergency treatment of asthma in relation to pollutant data from a six-site air sampling network, of which one station had monitoring for particles in the respirable range and for pollen. Specified air pollutant levels, including those for TSP, showed no significant geographic or temperature variation with respect to asthma outbreaks with the exception of small particles detected with special sampling that appeared to be significantly positively correlated with asthma admissions. However, few of the more obvious potentially confounding factors were evaluated or controlled for in the analysis.

Sterling et al. (1966, 1967) studied hospital admissions for "relevant" physiological causes in relation to measures of various air pollutants in Los Angeles. Significant decreases in respiratory symptom admissions were seen with decreasing SO_2, though SO_2 was low and varied little during the study. Other pollutants and weather may have been more important. How representative the pollutant data were of population exposures could not be determined based on scant information regarding monitor locations. Also, indications of considerable "adjustment" of raw data for health and pollution indices raise concerns regarding the validity of the reported findings.

A study in Philadelphia conducted by Girsh (1967) concluded that stagnant air conditions seemed to correlate with peak incidences of respiratory attacks. It is not clear how particles were measured or how representative the exposure data are for the subjects studied. Controls for day-of-week effects and medication usage were not discussed, but the effect of temperature was considered.

Cederlöf (1966) and Hrubec et al. (1973) studied chronic respiratory symptom prevalence in large panels of twins in Sweden and in the United States. An index of air pollution was used based on estimated residential and occupational exposures to SO_2, PM, and CO. Increased prevalence of respiratory symptoms in twins related to smoking, alcohol consumption, socioeconomic characteristics, and urban residence, but not to indices of air pollution. Ambiguities exist, however, regarding how representative the pollution index used would be of study population exposures and there was no clear delineation of symptom prevalence associations with SO_2 or PM levels specifically.

Verma et al. (1969) retrospectively studied absenteeism among New York City insurance workers over a 3-year period. They reported respiratory disease absenteeism to be correlated with SO_2 (controlled for temperature and season) but not significantly so after adjusting for time trends. No basis was provided by which to evaluate how representative the monitoring sites used were in relation to study population residences.

Winkelstein and Kantor (1969) conducted a survey of respiratory symptoms in a random sample of white women in Buffalo, New York. Among nonsmokers 45 years of age and over and smokers who did not change residence, respiratory symptoms were correlated with particulate matter concentrations obtained in the neighborhood of residence using a 21-station monitoring network. No association of symptom prevalence was seen with SO_2 concentrations. The authors expressed concern regarding the interviewing instrument used to collect health data, noting that it had not been standardized and had only a moderate degree of reproducibility.

Ishikawa et al. (1969) carried out comparisons of lungs obtained at autopsy from residents of St. Louis, Missouri and Winnipeg, Canada. Autopsy sets, matched for age, sex and race, showed more emphysema in the more polluted city (St. Louis). However, the autopsied groups may not accurately reflect the prevalence of disease in the general population, and familial and genetic factors were not taken into account. Nor were any means provided by which to credibly associate the health observations specifically with SO_2 or PM exposures.

Sulz et al. (1970) investigated hospitalizations for asthma (1956-1961) and for eczema (1951-1961) in Erie County, New York. Attack rates of patients were stratified into air pollution and social class categories. Area gradients in asthma and eczema hospitalization rates, adjusted for social class differences, were reported to correspond to the air pollution gradient. Meteorological differences between areas were not analyzed, and insufficient information was provided on pollution measurement methods.

Rao (1973) studied pediatric emergency room visits for asthma at Kings County Hospital, Brooklyn, N.Y., from October 1970 to March 1971, and reported a negative correlation of asthma visits with smoke shade (CoHs) levels. Lack of temperature adjustments and control for other confounding factors plus insufficient information concerning the representativeness of the exposure assessment, however, preclude acceptance of these findings.

Goldstein and Block (1974) similarly studied emergency room visits for asthma at a hospital in Harlem and one in Brooklyn, during September-December 1970 and September-December

1971. Temperature-adjusted asthma rates were positively correlated with SO_2 values in Brooklyn but not in Harlem. In the 1971 period, a 50-90 percent increase in asthma visits occurred on the 12 days of heaviest pollution. The size of the possible subject pool varied markedly between Brooklyn and Harlem, and the limited SO_2 data presented were below the 24-hour SO_2 standard.

Comstock et al. (1973) carried out a repeat survey in 1968-1969 of east coast telephone workers in New York, Philadelphia, and Baltimore and of telephone workers in Tokyo. After adjustment for age and smoking, no significant association of respiratory symptom prevalence was found with place of residence. However, no basis was provided by which to judge whether study population members either resided or lived within the limits of the city for which aerometric data were reported.

Speizer and Ferris (1973a,b) also conducted comparisons of respiratory symptoms and ventilatory function in central city and suburban Boston traffic policemen. These investigators found only slight, insignificant increases in symptom prevalence among nonsmokers and smokers, but not exsmokers, from the group in the more polluted central city area. No group differences in ventilatory function were found, however. The authors expressed caution regarding the findings, noting the small sample size, relatively limited exposure estimate data, and possibility of self-selection factors operating among the study population in choice of occupation and location of duty assignment.

Shy et al. (1973) examined the relationship between pulmonary function test (PFT) results for children in areas with different levels of pollutants in Cincinnati, Ohio. Appropriate pulmonary function data was gathered from over 300 students; and the TSP-and SO_2-exposure assessments are based upon data from a station within three blocks of the students' school. Most major potential confounding factors were also evaluated. Since acute effects of pollutant levels 24 hours prior to the PFT did not influence the performance of the children, the slight differences over the study period are difficult to ascribe to the small differences in then current TSP levels present at the time of the study. However, the potential contribution of previous pollutant exposures to the reported findings was not fully examined.

Mostardi and Martell (1975) evaluated pulmonary functions in junior high school students from a rural school district in comparison to a more polluted urban school district adjacent to Akron, Ohio. Pulmonary function levels were found to be lower among the urban students than the rural study group students, adjusted for height. Similar results were obtained by Mostardi and Leonard (1974) in a followup study of cohorts of the same students when in high school. Problems arguing for caution in attributing the observed differences to air pollution effects include self-selection of study populations via volunteering to participate, higher percentages of athletic participants likely having better lung function among the rural study population, and lack of more-than-cursory determination of SES factors. Also, the air pollution data reported (TSP and sulfation rate levels) were obtained from air monitoring sites located at the urban high school, up to 2-3 miles distant from student residences in the school district, and at a county site about 5 miles from the rural school complex and up to 10-13 miles from student residences in the rural school district.

Mostardi et al. (1981) similarly studied pulmonary function and respiratory disease in a group of grade school children in Akron, Ohio. They concluded that SO_2, in conjunction with NO_2, may be responsible for the observed pulmonary function decrements and symptoms. However, not enough information or discussion was provided to assess the adequacy of the aerometric data used, and no aerometric data were presented in this brief article.

Linn et al. (1976) studied respiratory symptoms and function in an office working population in Los Angeles and San Francisco in 1973. They found no significant difference in chronic respiratory symptom prevalence between cities; women in the more polluted community (Los Angeles) more often reported nonpersistent (< 2 years) production of cough and sputum. However, while the authors stated that they assumed the workers studied lived within close proximity to the single air monitor used in each city to yield aerometric data for the study, no information was provided by which to substantiate how representative the monitoring sites actually were in relation to population residences, and the study cannot be accepted as demonstrating clear no-effect levels at the TSP or other pollutant concentrations reported in the study.

A more recent study by Manfreda et al. (1978) was a qualitative study of urban-rural differences in two communities with similar air pollution concentrations and provides no clear quantitative information on the health effects of either SO_2 or PM.

Zagraniski et al. (1979) reported a cross-sectional investigation of two study groups instructed to report daily respiratory symptoms via postcard diaries in New Haven, Connecticut for 10 weeks in the summer of 1976. The two groups consisted of 192 telephone employees and 82 allergy and asthma clinic patients. The symptom rates were related to daily 8- and 24-hour ambient pollutant data recorded at two monitoring sites, 0.8 km from where the subjects were recruited. Suspended sulfate levels and TSP concentrations present in the ambient air at the time were found not to be statistically associated with acute adverse health effects, while oxidants appeared to be associated. Most major confounding factors, such as smoking and pollen levels, were examined. Several sources of bias, subject dropout, misuse of diaries, allergy sensitivity, and lack of indoor or personal monitoring were considered by the authors as unlikely to have biased their analysis. However, the study cannot be interpreted as demonstrating no-effects levels for sulfates or TSP due to the small study population and consequent low power for detecting small changes in health status likely to be occurring, if at all, at the air pollution levels reported.

Johnson et al. (1981) studied pulmonary function and symptoms in relation to pollutant levels in several Montana communities. Schoolchildren and chronic obstructive pulmonary disease (COPD) patients were administered pulmonary function tests and standardized questionnaires. Exposure estimates were based upon two to six monitoring sites in each community of the several studied. The adequacy of control for socioeconomic status or other possible confounding factors is not clear. The study does not show significance by some covariance or regression analyses but does by a sign test, making it difficult to determine the meaningfulness of the reported results at this time.

Detels et al. (1981) studied over 8,000 Los Angeles County residents' pulmonary function and respiratory symptoms in relation to ambient air pollutant levels. The results suggested that the lowest pulmonary function and greater number of symptoms occur most frequently in the area with high levels of photochemical oxidants and particles. The authors caution that these results are based on cross sectional data and note the limitation of the exposure assessment.

Ferris et al. (1979) reported on methodology being used in prospectively studying cohorts of 1500 to 1800 persons 25 to 75 years of age in each of six United States communities in regard to reported respiratory symptoms and measurements of lung function. Aerometric data (SO_2, NO_2, O_3, TSP, and mass respirable particles) are being gathered at centrally located stations and satellite stations. Additionally, some indoor/outdoor and personal monitoring for selected pollutants is being done. Ferris et al. (1980) reported preliminary cross-sectional data from this longitudinal study of the effects of pollutants on human health, which indicate that differences exist between geographic sites for the incidence of various respiratory symptoms studied and for age- and height-adjusted values for FEV. The data suggest trends toward positive associations of the observed effects with levels of pollution (including SO_2 and TSP) present in the different cities. However, Ferris et al. (1979) caution that any difference found between the cities in the cross-sectional study of adults might be attributable to past exposures or confounding factors such as differences in socioeconomic status. Further analysis of prospective data, still being collected, is therefore needed before definitive statements about the effects of exposure to specific levels of pollutants can be made based on these initial results of the Six Cities Study.

A series of studies was conducted by the U. S. Environmental Protection Agency under the Community Health and Environmental Surveillance System (CHESS) program, an integrated set of epidemiological studies performed between 1969 and 1975. In those studies, the health status of volunteer participants was either ascertained during single contacts or followed for time periods of up to nine months. Attempts were made to coordinate these health measures with air pollution observations from the residential neighborhoods of the study participants in an effort to derive information on quantatitive relationships between morbidity effects and both acute and chronic exposures to sulfur oxides, particulate matter and other air pollutants.

The results of approximately 10 CHESS studies were published in summary or review form in the early 1970s (Chapman et al., 1973; Hammer et al., 1976; Shy et al., 1973; French et al.,

1973) and were later presented in more detail in a 1974 EPA monograph entitled Health Consequences of Sulfur Oxides: A Report from CHESS 1970-1971, U.S. EPA document No EPA-650/1-74-004 (May, 1974). The manner in which the CHESS results were reported and interpreted in the 1974 monograph raised questions regarding inconsistencies in data collection and analyses, as well as possible mis- or overinterpretation of results for CHESS data sets discussed in the early published reports or the 1974 monograph.[a] Of particular concern with regard to many of the studies was the adequacy of aerometric data or other estimates of air quality parameters, as well as the collection and analysis of health endpoint measurement data, upon which numerous key conclusions were based regarding possible health effects-air pollution relationships. Many questions regarding the validity of most CHESS study findings published in the early 1970s still remain to be fully resolved, including the accuracy of data entries onto computer data tapes utilized in generating the analyses contained in the reports published in the early 1970s. In view of this, the potential usefulness of CHESS studies in yielding information on quantitative relationships between health effects and well-defined air concentrations of sulfur oxides and particulate matter is extremely limited at this time; such CHESS Program study results are, therefore, not included in the discussion in the main chapter text concerning morbidity effects.

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[a] The matter of misinterpretation or overinterpretation of data or results of analyses of data collected as part of the CHESS Program contributed to considerable controversy regarding the validity and accuracy of results of early CHESS studies, as interpreted and reported in a 1974 EPA monograph entitled Health Consequences of Sulfur Oxides: A Report from CHESS 1970-71, U.S. EPA Document No. EPA-6550/1-74-004 (May 1974). The controversy eventually led to the 1974 "CHESS Monograph" becoming the subject of U.S. Congressional oversight hearings in 1976. Subcommittees of the U.S. House of Representatives Committee on Science and Technology produced a report on the Monograph, other aspects of the CHESS Program, and EPA's air pollution research programs generally--a report entitled The Environmental Protection Agency's Research Program with Primary Emphasis on the Community Health and Environmental Surveillance System (CHESS): An Investigative Report. Of primary importance for the present discussion, that report, widely referred to either as the "Brown Committee Report" or the "Investigative Report" (IR), contained various comments regarding sources of error in CHESS Program air quality and health effects data and quality control problems associated with such data collection and analysis. The IR also contained various recommendations to be implemented by the Administrator of EPA pursuant to Section 10 of the Environmental Research, Development, and Demonstration Authorization Act of 1978 ("ERDDAA," P.L. 95-155, 91 Stat. 1275, November 8, 1977). ERDDAA also requires that EPA and the Agency's Science Advisory Board report to Congress on the implementation of the IR recommendations.

One recommendation of the IR was that an addendum to the 1974 sulfur oxides monograph be published, to be used in part to qualify the usefulness of the CHESS studies, and to apprise the public of the controversy surrounding CHESS. An addendum has been published, and is available from EPA, as announced in the Federal Register of April 2, 1980, 45 F.R. 21702. The addendum is incorporated by reference in this document in partial qualification of the CHESS studies cited herein and is part of the public file (or docket) established for revision of this criteria document. The Addendum contains the full text of the IR, reports to Congress by EPA on its implementation of the IR recommendations, and a report to Congress by EPA's Science Advisory Board on the same subject. The reader is directed to the Addendum and IR for detailed critiques of individual CHESS studies.

Several Canadian studies have investigated relationships between morbidity and chronic exposure to PM or SO_2, as described below.

Anderson and Larsen (1966) studied peak flowrates and school absence rates in children 6-7 years of age from three towns in British Columbia, Canada. Significant decreases in peak flowrates for children in two towns were reported to be associated with kraft pulp mill emissions. No effects on school absences were observed; however, ethnic differences were not studied.

Bates et al. (1962,1966) and Bates (1967) compared symptom prevalence, work absences, and ventilatory function in Canadian veterans residing in four Canadian cities. Lower prevalence of symptoms and work absences and better ventilatory function were found for veterans living in the least polluted city. However, it is not clear how representative the aerometric data were in relation to exposures of the study populations, based on lack of information regarding distance of the air monitoring sites from residences or work places of the subjects studied.

Bates (1973) also conducted a 10-year follow-up study of the Canadian veterans initially evaluated in 1960 and followed at yearly intervals with pulmonary function tests and clinical evaluations. The least decline in pulmonary function with age was seen for veterans from the least polluted city. Again, however, it is difficult to ascertain how representative any of the reported air pollution data were as estimates of study population exposures.

Another Canadian study (Neri et al., 1975) compared the prevalence of symptomatic disease and the level of pulmonary function in Sudbury, a mining community, and in Ottawa. Although chronic bronchitis was more prevalent in Sudbury men, 58 percent of Sudbury men had an occupational history suggesting high pollution exposure. Lung function levels were lower for both men and women in Sudbury. Very high periodic peak SO_2 exposure levels (exceeding 1000 $\mu g/m^3$) likely account more for any pollutant effects than long-term chronic exposures to relatively low annual average levels of SO_2 or annual mean particulate levels (which did not vary by much between Sudbury and Ottawa).

Becklake et al. (1978) compared the health status of children (9-11 years old) in three urban communities of Montreal, Canada with differing levels of air pollution. In the three areas studied, ambient SO_2 was reported to be 15, 123, and 59 $\mu g/m^3$, and annual mean high-volume TSP values were 84, 99, and 131 $\mu g/m^3$, respectively, for the low-, intermediate- and high-pollution areas, but there was a large overlap between areas. No significant differences were found in symptom prevalence, nor in overall pulmonary function measured by five lung function tests after controlling for socioeconomic factors. However, the analysis showed that one measurement, the closing volume (CC/TLC%), an index of small airway function, was consistently higher in children living in the two more polluted cites (one with high SO_2, the other with high TSP). In addition, there was a trend towards reduction in peak flowrates in the two more polluted cities compared to the third city. The authors speculated that these subtle differences might reflect the early stages of an irreversible obstructive airway disease, although follow-up studies would be necessary to test this.

A more recent study of the three communities in the Montreal area was marked by relatively small air pollution differences between cities (Aubrey et al., 1979). Cough and phlegm were weakly associated with air pollution concentration, but lung function was not. However, after accounting for intercity differences in age and smoking, none of the associations were statistically significant. Little meaningful quantitative information can be extracted from the report, and it is difficult to evaluate the representativeness of the aerometric data reported in relation to study population exposures.

APPENDIX 14B

OCCUPATIONAL HEALTH STUDIES ON PARTICULATE MATTER AND SULFUR OXIDES

Armstrong, B. K., J. C. McNulty, L. J. Levitt, K. A. Williams, and M. S. T. Hoffs. Mortality in Gold and Coal Miners in Western Australia with Special Reference to Lung Cancer. Brit. J. Ind. Med. 36:199-205, 1979.

Axelson, O., E. Dahlgren, C. D. Jansson, and S. O. Rehnlund. Arsenic Exposure and Mortality: A Case-Referent Study from a Swedish Copper Smelter. Brit. J. Ind. Med. 35:8-15, 1978.

BarZiv, J. and G. M. Goldberg. Simple Siliceous Pneumoconiosis in Negev Bedouins. Arch. Environ. Health 29:121-126, 1974.

Bennett, J. G., J. A. Dick, Y. S. Kaplan, P. A. Shand, D. H. Shennan, D. J. Thomas, and J. S. Washington. The relationship between coal rank and the prevalence of pneumoconiosis. Brit. J. Ind. Med. 36:206-210, 1979.

Bernard, T. E., E. Kamon, and R. L. Stein. Respiratory Responses of Coal Miners for Use with Mechanical Simulators. AIHAJ 39:425-429, 1978.

Bonnevie, A. Silicosis and Individual Susceptibility--Fact or Myth? Ann. of Occup. Hyg. 20:101-108, 1977.

Brambilla, C., J. Abraham, E. Brambilla, K. Benirschke, and C. Bloor. Comparative Pathology of Silicate Pneumoconiosis. Amer. J. Path. 96:149-163, 1979.

Carlson, M. L., and G. R. Peterson. Mortality of California Agricultural Workers. J.O.M. 20:30-32, 1978.

Chan-Yeung, M., M. Schulzer, L. MacLean, E. Dorken, F. Tan, D. Enarson, and S. Grzybowski. A follow-up Study of the Grain Elevator Workers in the Port of Vancouver. Amer. Rev. Resp. Dis. 121:228, 1980. (Supplement).

Chan-Yeung, M., M. Schulzer, L. MacLean, E. Dorken, and S. Grzybowski. Epidemiologic Health Survey of Grain Elevator Workers in British Columbia. Amer Review Resp. Disease 121:329-338, 1980.

Corey, P., I. Broder, and M. Hutcheon. Relationship Between Dust Exposure of Grain Elevator Workers and Both Baseline Pulmonary Function and Acute Work-Related Changes in Status. Amer. Rev. Resp. Dis. 121:228, 1980 (Supplement).

Corn, M., Y. Hammad, D. Whittier, and N. Kotski. Employee Exposure to Airborne Fiber and Total Particulate Matter in Two Mineral Wool Facilities. Environ. Res. 12:59-74, 1976.

Craighead, J. E., and N. V. Vallyathan. Cryptic pulmonary lesions in workers occupationally exposed to dust containing silica. JAMA 244:1939-1941, 1980.

Crosbie, W. A., R. A. F. Cox, J. V. Leblanc, and D. Cooper. Survey of Respiratory Disease in Carbon Black Workers in the U.K. and U.S.A. Amer. Rev. Resp. Dis. 119, 1979 (Supplement).

Decoufle, P., and D. J. Wood. Mortality Patterns Among Workers in a Gray Iron Foundry. Amer. J. Epidemiol. 109:667, 1979.

Dosman, J. A., D. J. Cotton, B. L. Graham, K. Y. Robert, F. Froh, and G. D. Barnett. Chronic Bronchitis and Decreased Forced Expiratory Flow Rates in Lifetime Nonsmoking Grain Workers. Amer. Review Resp. Disease 121:11-16, 1980.

Dutkiewicz, J. Exposure to Dust-Borne Bacteria in Agriculture. I. Environmental Studies. Arch. Envr. Health. 33:250-259, 1978.

Dutkiewica, J. Exposure to Dust-Borne Bacteria in Agriculture. II. Immunological Survey. Arch. Envr. Health. 33:260-270, 1978.

Farant, J., and C. F. Moore. Dust Exposures in the Canadian Grain Industry. AIHAJ 39:177-194, 1978.

Federspiel, C. F., J. T. Layne, C. Auer, and J. Bruce. Lung Function Among Employees of a Copper Mine Smelter: Lack of Effect of Chronic Sulfur Dioxide Exposure. J.O.M. 22:438-444, 1980.

Ferris, B. G., S. Puleo, and H. Y. Chen. Mortality and morbidity in a pulp and a paper mill in the United States: A ten-year follow-up. Brit. J. Ind. Med. 36:127-134, 1979.

Gibson, E. S., R. H. Martin, and J. N. Lockington. Lung Cancer Mortality in a Steel Foundry. J.O.M., 19:807-12, 1977.

Glover, J. R., C. Bevan, J. E. Cotes, P. C. Elwood, N. G. Hodges, R. L. Kell, C. R. Lowe, M. McDermott, and P. D. Oldham. Effects of Exposure to Slate Dust in North Wales, Brit. J. Ind. Med. 37:152-162, 1980.

Gold, A., W. A. Burgess, and E. V. Clougherty. Exposure of Firefighters to Toxic Air Contaminants. AIHAJ 39:534-539, 1978.

Jedrychowski, W. A Consideration of Risk Factors and Development of Chronic Bronchitis in a Five-Year Follow-up Study of an Industrial Population. J. Epidemio. and Comm. Health 33:210-214, 1979.

Karol, M. H., H. H. Ioset, and Y. C. Alarie. Effect of Coal Dust Inhalation on Pulmonary Immunologic Responses. AIHAJ 40:284-290, 1979.

Klosterkotter, W. New Aspects on Dust and Pneumoconiosis Research. AIHAJ 36:659-668, 1975.

Kung, V. A. Morphological Investigations of Fibrogenic Action of Estonian Oil Shale Dust. Environ. Health Perspect. 30:155-156, 1979.

Lapp, N. L., J. L. Hankinson, D. B. Burgess, and R. O'Brien. Changes in Ventilatory Function in Coal Miners After a Work Shift. Arch. Environ. Health. 24:204-208, 1972.

Lowe, C. R., H. Campbell, and T. Kosha. Bronchitis in Two Integrated Steel Works. III. Respiratory Symptoms and Ventilatory Capacity Related to Atmospheric Pollution. Br. J. Industr. Med. 27:121-29, 1970.

Lowe, C. R., P. L. Pelmear, H. Campbell, R. A. N. Hitchens, T. Khosla, and T. C. King. Bronchitis in Two Integrated Steel Works. I. Ventilatory Capacity, Age, and Physique of Non-Bronchitic Men. Br. J. Prev. Soc. Med., 22:1-11, 1968.

Milham, S. Mortality in Aluminum Reduction Plant Workers. J. Occup. Med., 21:475-480, 1979.

Morgan, W. K. C. Industrial Bronchitis. Brit. J. Ind. Med. 35:285-291, 1978.

Morgan, W. K. C. Magnetite Pneumoconiosis. J.O.M. 20:762-763, 1978.

Musk, A. W., J. M. Peters, D. H. Wegman, and L. J. Fine. Pulmonary Function in Granite Dust Exposure: A Four-Year Follow-up. Amer. Review Resp. Disease 115:769-776, 1977.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Beryllium. DHHS (NIOSH) 72-10268, U. S. Department of Health and Human Services, Washington, DC, 1972.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Coke Oven Emissions. DHHS (NIOSH) 73-11016, U. S. Department of Health and Human Services, Washington, DC, 1973.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Inorganic Mercury. DHHS (NIOSH) 73-11024, U. S. Department of Health and Human Services, Washington, DC, 1973.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Sulfuric Acid. DHHS (NIOSH) 74-128, U. S. Department of Health and Human Services, Washington, DC, 1974.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Ammonia. DHHS (NIOSH) 74-136, U. S. Department of Health and Human Services, Washington, DC, 1974.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Cotton Dust. DHHS (NIOSH) 75-118, U. S. Department of Health and Human Services, Washington, DC, 1975.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Inorganic Arsenic. (Revised). DHHS (NIOSH) 75-149, U. S. Department of Health and Human Services, Washington, DC, 1975.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Zinc Oxide. DHHS (NIOSH) 76-104, U. S. Department of Health and Human Services, Washington, DC, 1974.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Chromium VI. DHHS (NIOSH) 76-129, U. S. Department of Health and Human Services, Washington, DC, 1976.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Nitric Acid. DHHS (NIOSH) 76-141, U. S. Department of Health and Human Services, Washington, DC, 1976.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Cadmium. DHHS (NIOSH) 76-192, U. S. Department of Health and Human Services, Washington, DC, 1976.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Hydrogen Cyanide and Cyanide Salts. DHHS (NIOSH) 77-108, U. S. Department of Health and Human Services, Washington, DC, 1977.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Organotin Compounds. DHHS (NIOSH) 77-115, U. S. Department of Health and Human Services, Washington, DC, 1977.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Inorganic Nickel. DHHS (NIOSH) 77-164, U. S. Department of Health and Human Services, Washington, DC, 1977.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Asbestos. DHHS (NIOSH) 77-169, U. S. Department of Health and Human Services, Washington, DC, 1977.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Vanadium. DHHS (NIOSH) 77-222, U. S. Department of Health and Human Services, Washington, DC, 1977.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Tungsten and Cemented Tungsten Carbide. DHHS (NIOSH) 77-227, U. S. Department of Health and Human Services, Washington, DC, 1977.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Coal Tar Products. DHHS (NIOSH) 78-107, U. S. Department of Health and Human Services, Washington, DC, 1978.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Inorganic Lead. DHHS (NIOSH) 78-158, U. S. Department of Health and Human Services, Washington, DC, 1978.

National Institute for Occupational Safety and Health. Criteria for a recommended standard: Occupational Exposure to Carbon Black. DHHS (NIOSH) 78-204, U. S. Department of Health and Human Services, Washington, DC, 1978.

Rencher, A. C., M. W. Carter, and D. W. McKee. A Retrospective Epidemiological Study of Mortality at a Large Western Copper Smelter. J.O.M. 19:754-58, 1977.

Robertson, J. McD, and T. H. Ingalls. A Mortality Study of Carbon Black Workers in the United States from 1935 to 1974. Arch. Environ. Health 35:181-186, 1980.

Rockette, H. E. Cause Specific Mortality of Coal Miners. J.O.M. 19:795-801, 1977.

Saric, M., I. Kalacic, and A. Holetic. Follow-up of Ventilatory Lung Function in a Group of Cement Workers. Brit. J. Inc. Med. 33:18-24, 1976.

Sherwin, R. P., M. L. Barman, and J. L. Abraham. Silicate Pneumoconiosis of Farm Workers. Lab. Invest. 40:576-582, 1979.

Sweet, D. V., W. E. Crouse, J. V. Crable, J. R. Carlberg, and W. S. Lainhart. The Relationship of Total Dust, Free Silica, and Trace Metal Concentrations to the Occupational Respiratory Disease of Bituminous Coal Miners. AIHAJ 35:479-488, 1974.

Valic, F., D. Beritic-Stahuljak, and B. Mark. A Follow-Up Study of Functional and Radiological Lung Changes in Carbon-Black Exposure. Int. Arch. Occup., 34:51-63, 1975.

Warren, C. P. W. Lung Disease in Farmers. Cand. Med. Assn. J. 116:391-394, 1977.

Whiting, W. B. Occupational Illnesses and Injuries of California Agricultural Workers. J.O.M. 17:177-181, 1975.

APPENDIX 14C

SUMMARY OF EXAMPLES OF SOURCES AND MAGNITUDES OF MEASUREMENT ERRORS ASSOCIATED WITH AEROMETRIC MEASUREMENTS OF PM AND SO_2 USED IN BRITISH AND AMERICAN EPIDEMIOLOGICAL STUDIES

TABLE 14C-1. SUMMARY OF EVALUATION OF SOURCES, MAGNITUDES, AND DIRECTIONAL BIASES OF ERRORS ASSOCIATED WITH BRITISH SO_2 MEASUREMENTS

Time period	Measurement method	Reported source of error	Direction and magnitude of reported error	Likely general impact on British SO_2 data
Pre-1961	Lead Dioxide	Humidity (RH)	Reaction rate increases with RH.	Variable positive bias, especially in summer.
		Temperature (T)	Reaction rate increases 2% per 5° rise.	Variable positive bias, especially in summer.
		Wind speed (WS)	Reaction rate increases with WS.	Variable positive bias, under high wind cond.
		(Overall errors)		Can be up to ± 180 $\mu g/m^3$ (2σ).
1961-1980 (British National Air Pol. Survey)	Hydrogen Peroxide	Siting of Sample Line Intake:		
		a. too near boiler chimneys	50 - 100 $\mu g/m^3$ overestimation.	Occasional (prob. rare) positive bias.
		b. too near vegetation	50 - 70 percent underestimation.	Occasional (prob. rare) negative bias.
		Sample Line Adsorption:		
		a. Good care & cleaning	10 $\mu g/m^3$ underestimation.	Possible general 10 $\mu g/m^3$ negative bias.[a]
		b. Average care	20-25 $\mu g/m^3$ low from 50 $\mu g/m^3$.	Occasional 40-50% negative bias.
		c. Poor care (insects, dirt)	Probable greater underestimation.	Likely rare 50-90% negative bias.
		Flow Meter Problems:		
		a. Daily normal conditions	± 3 percent variation.	Negligible impact. Presumed ±3% precision of data.
		b. 8-port unit with only one weekly flow reading	± 5 percent variation.	-5% negative bias on high SO_2-BS days. +5% positive bias on low SO_2-BS days.
		Allowable Filter Clamp Leakage	1-2 percent underestimation.	Negligible impact. Presumed ±2% precision of data.
		Poor Clamp Care & Technique	5-10 percent underestimation.	Likely occasional 5-10% negative bias.
		Grade B Glassware Usage	2-5 $\mu g/m^3$ underestimation.	Negligible impact.
		Improper Alkalinity Buffering	5-10 $\mu g/m^3$ underestimation.	Occasional 5-10 $\mu g/m^3$ negative bias.
		CO_2 in Demineralized in H_2O	40 $\mu g/m^3$ low from 50 $\mu g/m^3$ monthly mean.	Occasional negative bias of up to 80%.[b]
		Atmospheric Ammonia	25 $\mu g/m^3$ underestimation on 10% of summer samples in urban areas. $\leq$80 $\mu g/m^3$ low ind. days & 40 $\mu g/m^3$ low monthly summer mean in country areas.	$\geq$25 $\mu g/m^3$ neg. bias on 10% of summer samples in urban areas. Occasional neg. bias in country areas- up to 80 $\mu g/m^3$ daily data & up to 100% monthly mean in summer.
		Titration Error:		
		a. Normal-sharp color change of indicator at pH 4.5	±5 $\mu g/m^3$ error in determination.	Presumed ± 5 $\mu g/m^3$ precision of data.
		b. Gradual color change of indicator at pH 4.5	±10 $\mu g/m^3$ error in determination.	Actual ± 10 $\mu g/m^3$ precision level.[c]
		c. Rounding off to 0.1 ml of alkali volume added.	±5 $\mu g/m^3$ error in determination.	Added ± 5 $\mu g/m^3$ precision error.[c]
		Evaporation of reagent:	$\leq$15 $\mu g/m^3$ over estimation, especially in summer months.	15-100% pos. bias for SO_2 data <100 $\mu g/m^3$. 7.5-15% pos. bias for SO_2 of 100-200 $\mu g/m^3$. 3.25-7.5% pos. bias for SO_2 of 200-400 $\mu g/m^3$. $\leq$3.25% pos. bias for SO_2 data >400 $\mu g/m^3$.
		Temperature and Pressure:		
		a. Corrections - normal	5% underestimation.	General 5% neg. bias in SO_2 data.
		b. Large ΔP at filter	10% underestimation.	Occasional ~ ±10% negative bias in SO_2 data.

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[a] Data from 1965-1968 most clearly impacted.

[b] Data from 1966-1967 most clearly impacted.

[c] At $\leq$50 $\mu g/m^3$ uncertainty due to these two errors is ~ 7 $\mu g/m^3$ or 14%. That is, 68% of the data are within 14% and 5% are >28% in error.

TABLE 14C-2. SUMMARY OF EVALUATION OF SOURCES, MAGNITUDES, AND DIRECTIONAL BIASES OF ERRORS ASSOCIATED WITH BRITISH SMOKE (PARTICULATE) MEASUREMENTS

Time period	Measurement method	Reported source of error	Direction and magnitude of reported error	Likely general impact on published BS data
1944-1950s	Smoke filter	Leakage at clamp. Weights used to make the seal.	Highly variable underestimation of BS levels.	Probable widespread highly variable negative bias.
Pre-1961		Comparing reflectance to photographs of painted standard stains.	Depending upon observer and value of R.	Probable widespread relatively small negative bias.
		Reflectance (R) below 25%, stain too dark with use of Clark-Owens DSIR curve.	50-100% underestimation.	Occasional 50-100% negative bias in some data sets.
1961-1964		Computer not following proper calibration curve.	≤80% underestimation at low R if not corrected by WSL (See Moulds,1961) and discussion of clamp size correction factor.	Negligible for BS <~100 μg/m^3. Increasing negative bias up to 80% as BS values increase over 100 μg/m^3.
1964-1980		Clamp correction factor for other than 1-inch clamp.	Uncertain; derivation cannot be verified. Possible ±20%.	Possible underestimate for 2-inch and 4-inch clamps Possible overestimate for 1/2-inch and 10 cm clamps.
		Flow rate - normal 1 day. Flow by 8-port with 1 reading per week.	±3% variation. -10% underestimation. +10% overestimation.	Presumed ± 3% precision level. 10% negative bias on high BS days. 10% positive bias on low BS days.
		Variability of reading reflectance.	±2 units of R	Error increases with BS level from ±10% at 50 μg/m^3 up to ±20% at 400 μg/m^3.
		Averaging reflectance instead of averaging mass/cm^2.	Highly variable underestimation due to nonlinearity of R.	Probable small negative bias at low BS levels, could be large at high BS.
		Use of coarse side of filter facing upstream.	6-15% underestimation.	Occasional negative bias of 6-15%.

TABLE 14C-2 (continued).

Time period	Measurement method	Reported source of error	Direction and magnitude of reported error	Likely general impact on published BS data
		Reading of wrong side of stained filter.	50-75% underestimation.	Occasional negative bias of 50-75%.
		Leakage at filter clamp		
		a. Normal, with good care	1-2% underestimation.	General 1-2% negative bias.
		b. With inadequate care.	2-8% underestimation.	Occasional 2-8% negative bias.
		c. Careless loading where uneven stains are produced.	10-20% underestimation.	Occasional 10-20% negative bias
		Use of wrong clamp size		
		a. Stain too light R>90%.	Highly variable overestimation.	Data usage not recommended.
		b. Stain too dark R<25%.	Highly variable underestimation.	Data usage not recommended.

TABLE 14C-3. SUMMARY OF EVALUATION OF SOURCES, MAGNITUDES, AND DIRECTIONAL BIASES OF ERRORS ASSOCIATED WITH AMERICAN SO_2 MEASUREMENTS

Time period	Measurement method	Reported source of error	Direction and magnitude of reported error	Likely general impact on American SO_2 data
1944-1968	Lead dioxide.	Humidity (RH).	Reaction rate increases with RH.	Variable positive bias, especially in summer.
		Temperature (T).	Reaction rate increases 2% per 5° rise.	Variable positive bias, especially in summer.
		Windspeed (WS).	Reaction rate increases with WS.	Variable positive bias, especially in summer.
		Saturation of Reagent (sulfation plate mainly).	Variable underestimation beyond pt. where 15% of PbO_3 on plate reacted.	Possible large negative bias, especially for 30-day samples for summer monthly readings.
		(Overall Errors).		Generally wide ± error band associated with data. Possible negative bias up to ≥100%, mainly in summer, with 30-day reading.
1969-1975 (EPA CHESS PROGRAM)	West-Gaeke Pararosanaline.	Spillage of reagent during shipment.	18% of total volume 50% of time; occasional total loss	Half of SO_2 data likely negatively biased by mean of 17%; some up to 100%.[a]
		Time delay for reagent-SO_2 complex.	SO_2 losses of 1.0, 5, 25, and 75% at 20, 30, 40, and 50°C, respectively.	Usually small (<5%) negative bias, but consistent negative summer bias up to 25% at 40°C temp, extreme.[b]
		Concentration dependence of sampling method.	Underestimation of unspecified magnitude at daily SO_2 >200 $\mu g/m^3$.	Probable general negative bias in daily, monthly, and yearly SO_2 data.[b]
		Low flow correction.	±10% to 50% variable error.	Usually error of < ±10%; occasionally up to ± 50% in daily, but dampened statistically in annual mean.[b]
		Bubbler train leakage.	Small underestimation air of unspecified magnitude.	Slight negative bias suspected.[b]
		(Overall errors).		From Nov., 1970, to Dec., 1971, data biased low by 50-100%.[c] From Nov., 1971, to conclusion of CHESS Program in 1975, fall-winter data appear valid but summer data biased low by maximum of 60-80%.[c] From 1972 to 1975 annual average data approximately 15-20% low. Daily data highly random, not useful.[c]

[a] November, 1970, to April, 1973, CHESS Program data impacted before error corrected.
[b] Applies to CHESS Program SO_2 data from all years 1970-1975.
[c] As summarized by Congressional Investigative Report (IR, 1976).

TABLE 14C-4. SUMMARY OF EVALUATION OF SOURCES, MAGNITUDES, AND DIRECTIONAL BIASES OF ERRORS ASSOCIATED WITH AMERICAN TOTAL SUSPENDED PARTICULATE (TSP) MEASUREMENTS

Time period	Measurement method	Reported source of error	Direction and magnitude of reported error	Likely general impact on published TSP data
1954-1980	Staplex Hi Vol TSP	Time Off (Due to power failure).	Variable underestimation.	Negligible impact, rare negative bias.
		Weighing error.	±2% random variation.	Negligible impact.
		Flow measurement (with control).	±2% random variation.	Negligible impact.
		Flow measurement (without control).		
		a. Constant TSP--Average of flows.		
		1. Low TSP level.	2% underestimation.	Negligible impact.
		2. High TSP level.	5-10% underestimation.	Possible 5-10% negative bias.
		b. Rising TSP-Average of flows.	10-20% underestimation.	Possible 10-20% negative bias.
		c. Falling TSP-Average of flows.	10-20% overestimation.	Possible 10-20% possible bias.
		Aerosol evaporation on standing.	1-2% underestimation.	Probable negligible impact.
		Condensation of water vapor.	5% overestimation.	Possible 5% positive bias.
		Foreign bodies on filter (Insects).	Generally small overestimation.	Possible 5% positive bias.
		Windblown dust into filter during off-mode.	Generally small overestimation.	Occasional (rare) positive bias.
		Wind speed effect on penetration of dust into the Hi-Vol shelter.	Less penetration at high windspeed.	Occasional (rare) negative bias.
		Wind direction effect due to Hi-Vol Asymmetry.	Higher penetration when normal to sides.	Probable increase in random (±) error.
		Artifact formation, NO_3^- $SO_4^=$.	5-10 μg/m^3 overestimation.	Occasional positive bias.

TABLE 14C-4. (continued)

Time period	Measurement method	Reported source of error	Direction and magnitude of reported error	Likely general impact on published TSP data
1969-1975 (EPA CHESS Program).	Fed. Reference Method Standard Hi-Vol Sampler	Loss of sampling material in field.	No specific estimate of magnitude of error; but would be underestimation.	Probable slight negative bias in Utah winter data. No known impact on other CHESS TSP data.
		Loss of sampling material in mailing.	Reported 4-25% apparent loss; max. likely due to crustal (sand, etc.) fall-off from selected Utah sampling sites.	Probable general small <10% negative bias; occasional 25% negative bias.
		Evaporation of organic substances.	No specific estimate of error magnitude, but not likely to exceed 5% underestimation.	Probable slight negative bias of <5% for TSP data from urban/industrial areas.
		Windflow velocity and asymmetry.	No specific estimate of error magnitude; but most likely to increase random variation of small underestimation.	Negligible impact or slight negative bias.
		(Overall errors).		Generally <10% negative bias; occasional 10 to 30% negative bias.

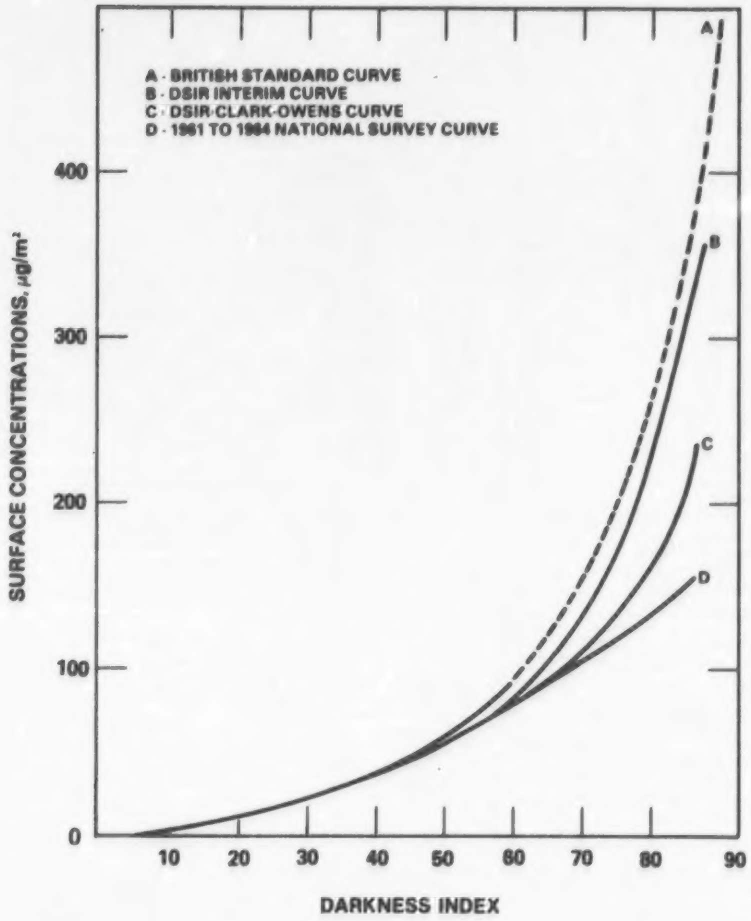


Figure 14C-1. Comparison of smoke calibration curves for Eel reflectometer, Whatman No. 1 paper, and a 1-in.-diameter filter. The computer followed curve D during 1961-64 instead of the correct curve(s) B and C. All British epidemiology studies using BS ($\mu g/m^3$) data computed by WSL from reflectance readings for United Kingdom communities during 1961-64 period are, therefore, open to criticism on the basis of utilizing and reporting erroneous BS aerometric data. Due to the non-linear relationships involved, no simple correction factor can be used by which to estimate the 'correct' BS values in $\mu g/m^3$ from values reported in such studies.
Source: Warren Spring Laboratory (1967).

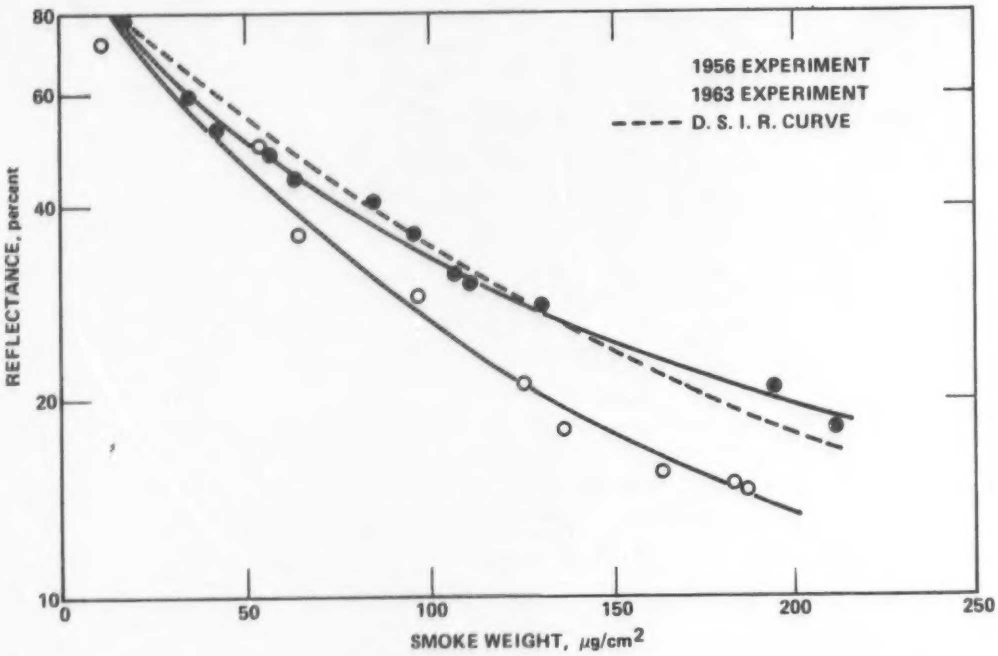


Figure 14C-2. Relationship between reflectance (log scale) and weight of smoke per unit area.

Source: Waller (1964).

APPENDIX 14D

EPA REANALYSIS OF MARTIN AND BRADLEY (1960) DATA ON MORTALITY DURING 1958-59 LONDON WINTER

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

DATE September 25, 1981

SUBJECT **Reanalysis of the Martin and Bradley Mortality Data**

FROM Victor Hasselblad Vic Hasselblad
Andrew G. Stead Andrew G. Stead
Biometry Division (MD-55)

TO Lester Grant
Director, ECAO (MD-52)

We attempted to reanalyze the Martin and Bradley data in a manner that would not only describe the relationship of mortality to smoke, but would also give some estimates of the variability involved. We decided to fit a nonlinear curve to the data, using the form

$$Y = a + \exp(b + c \log X).$$

This form was chosen because it allows for a lower asymptote (a), and the curve can rise sharply if necessary. The shape of the curve is similar to the one found by Schimmel. Since deaths are counts we tried the classical square root transformation, and found that this did improve the fit. We did not use the 15 day moving average, nor did we use any other seasonal or trend adjustment. We eliminated the month of February in order to minimize the effect of influenza.

The data along with the fitted curve are in Figure 1. The estimates of a, b, and c were 16.73, .169, and 1.58 respectively. The estimated asymptotic background rate is $(16.73)^2$ or 280 deaths per day.

The 95 percent confidence intervals can be estimated using the formula (Kendall and Stuart, Vol. 1, p. 247) for the variance of a function:

$$\mathrm{Var}(g(\underline{\theta})) \doteq \left(\frac{\partial g}{\partial \underline{\theta}}\right)' V \left(\frac{\partial g}{\partial \underline{\theta}}\right), \text{ where}$$

V is the estimated asymptotic covariance matrix of parameter estimates, and

$$\frac{\partial g}{\partial \underline{\theta}} = \begin{pmatrix} \frac{\partial g}{\partial a} \\ \frac{\partial g}{\partial b} \\ \frac{\partial g}{\partial c} \end{pmatrix} \quad \text{in our case.}$$

These estimated 95 percent confidence limits are also in Figure 1.

All of the above calculations are based on the assumption the dose variable is measured exactly for each person, and without error. Since there is both variation and measurement error in the dose (smoke) variable, we attempted to get some very crude estimates of this. By making the assumption that each of the 7 monitoring stations were located at "random" points throughout the city of London, we calculated the variances and standard deviations for each day. We found that the standard deviations increased approximately linearly with the mean, as shown in Figure 2. Because of this, and because of other work with smoke data, we based our calculations on the logarithms of the smoke data. The average variance of the logarithms was .0802, giving a geometric standard deviation of 1.33. This means that an approximate 95 percent confidence interval for a single day's average of 7 readings would be given by multiplying and dividing the geometric mean by

$$\exp(1.96(.0802/7)^{1/2}) = 1.233.$$

The next step in the calculations is extremely crude. We used this range of dose values to recalculate the 95 percent confidence interval for the curve in Figure 1. The lower limits came from using X/1.233 instead of X in the formula for the confidence interval. The upper limits came from using 1.233 X in the formula. The resulting broader band about the curve is shown in Figure 3.

Several important caveats must be mentioned. All of the calculations are based on approximations and assumptions of unknown accuracy. Thus the resulting curve may be little better than a best guess. Many contributing factors should be considered and additional analysis techniques should be investigated before these calculations could be considered scientifically sound. The calculations represent the best estimate we can make in a very limited amount of time.

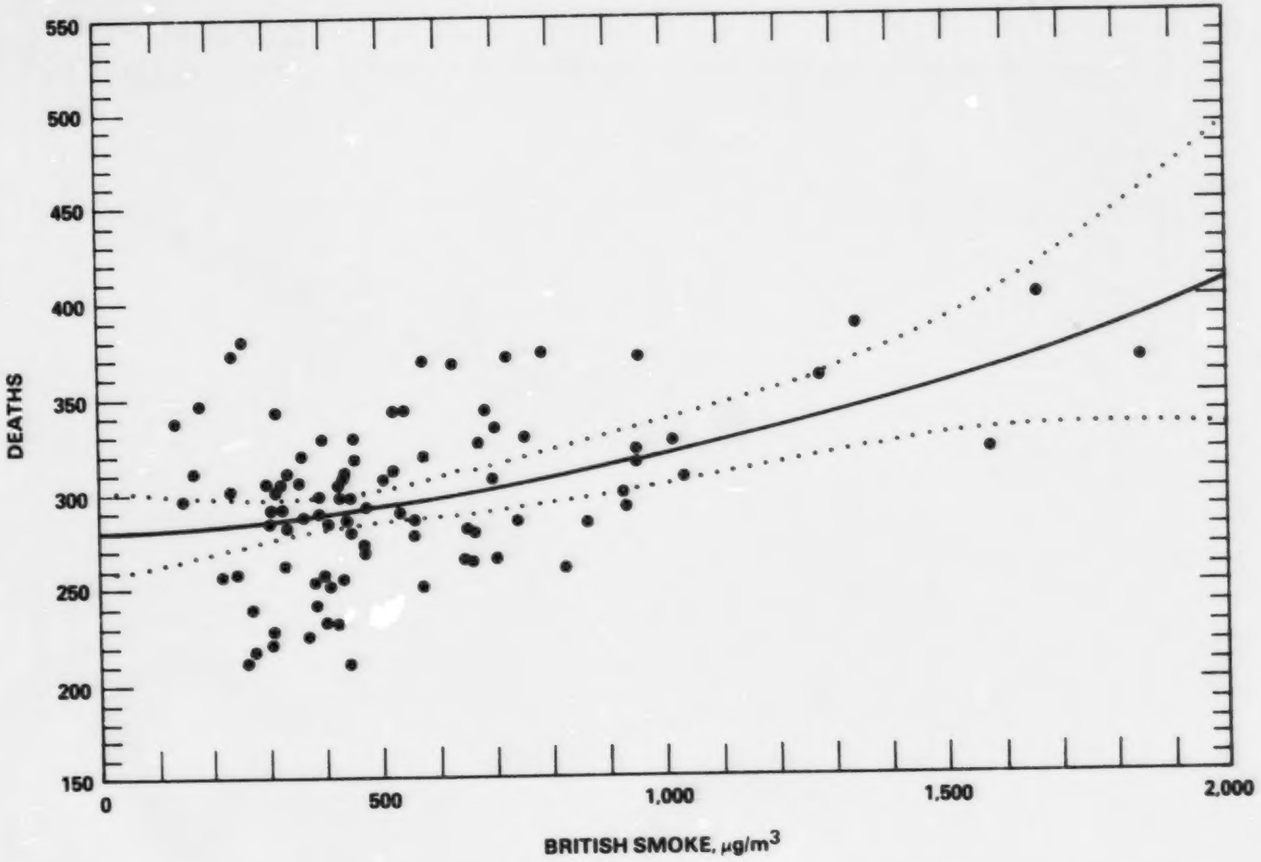


Figure 14D-1. Fitted curve and 95 percent confidence interval for data of Martin and Bradley (vertical axis changed), assuming error in mortality but not in smoke data.

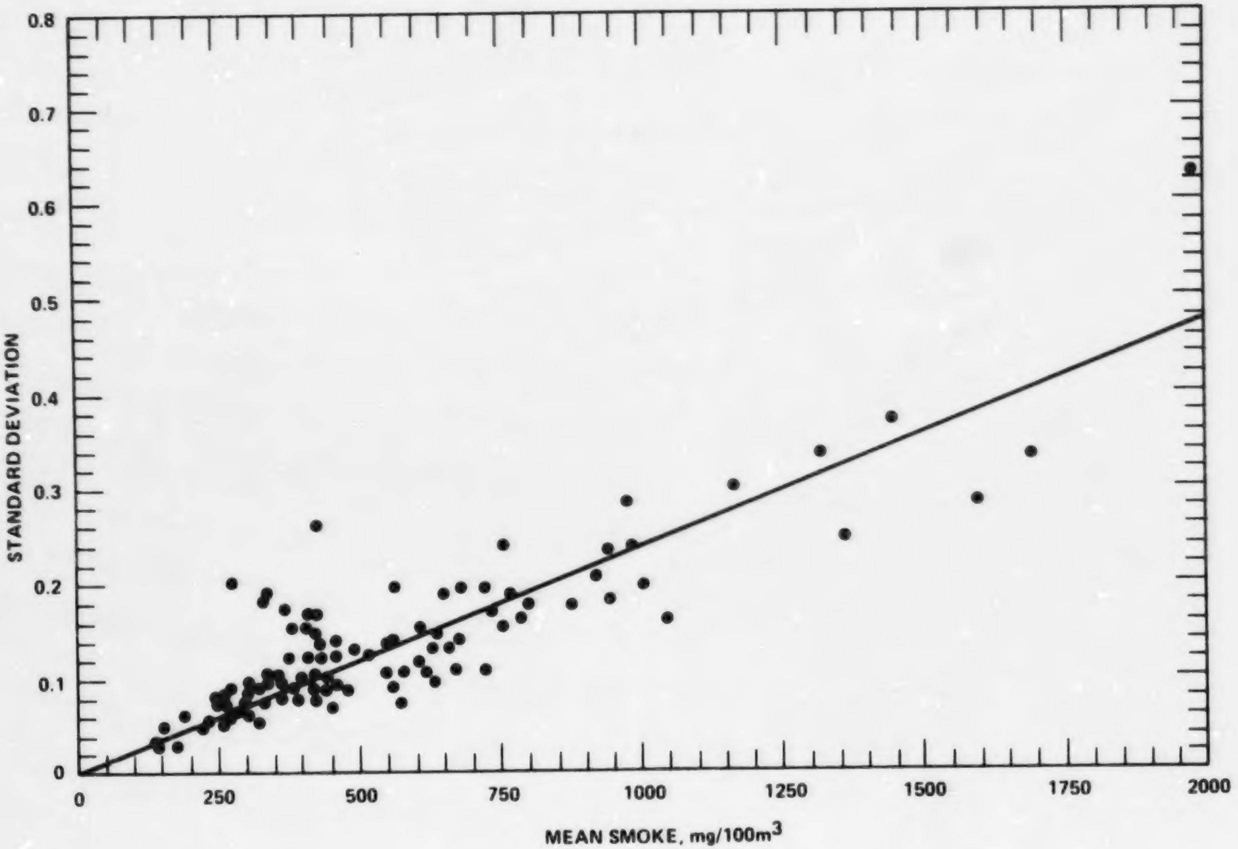


Figure 14D-2. Relationship of standard deviation to mean for smoke data of Martin and Bradley.

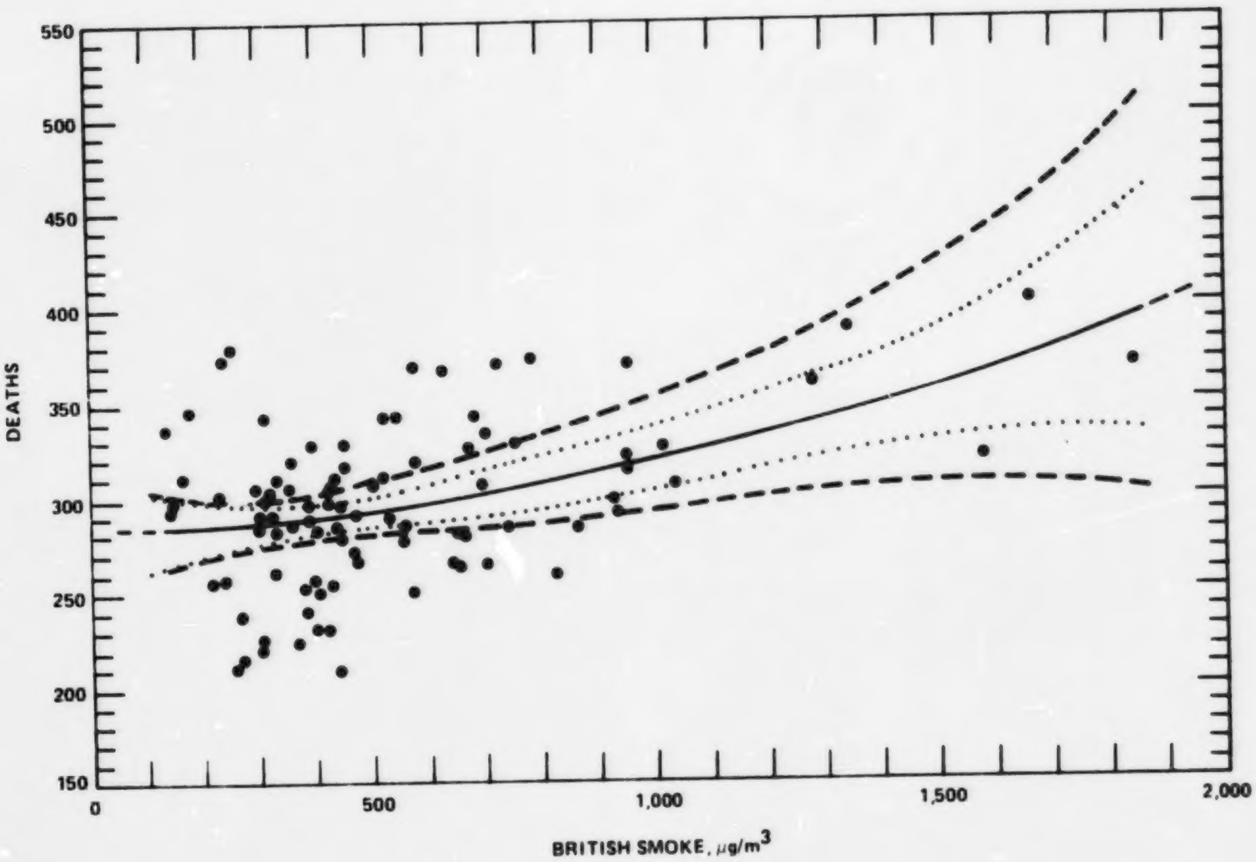


Figure 14D-3 Fitted curve (———) depicting dose-response relationship between mortality (number of daily deaths) and atmospheric particulate matter concentrations (expressed in $\mu g/m^3$ nominal British Smoke) during London winter of 1958/59, as determined by EPA reanalysis of Martin and Bradley (1960) data. Also shown are: 95% confidence intervals (..........) assuming error in mortality data but not smoke data; and 95% condidence intervals (— — —) taking into account variation or error in both mortality and smoke data.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

DATE October 2, 1981

SUBJECT Continued Reanalysis of the Martin and Bradley Mortality Data

FROM Victor Hasselblad
Andrew G. Stead
Biometry Division (MD-55) Vic Hasselblad

TO Lester Grant, Director
Environmental Criteria and Assessment Office (MD-52)

We performed the same analyses on the SO_2 data of Martin and Bradley as we had with the smoke data. As with the smoke data, we found that the log transformation of the SO_2 levels appeared to be the most appropriate. The average variance of the logarithms of the SO_2 data was .1725. Thus the 95-percent confidence intervals for SO_2 city wide means are given by multiplying and dividing the geometric means by:

$$\exp(1.96(.1725/7)^{1/2}) = 1.360$$

The graphs for the fitted mortality data are in figures 1, 1A, 3, and 3A. The model in this case became

$$(Y)^{1/2} = 16.94 + \exp(1.73 + 1.47 \log(SO_2))$$

We would like to again state that these calculations are crude. Many factors have not been considered. For example, we did not use 15-day moving averages as did Martin and Bradley. We do believe that the figures give some feeling for the inherent variability in the data.

One minor correction should be made in the smoke graphs. The curves should not be extended beyond the range of the data.

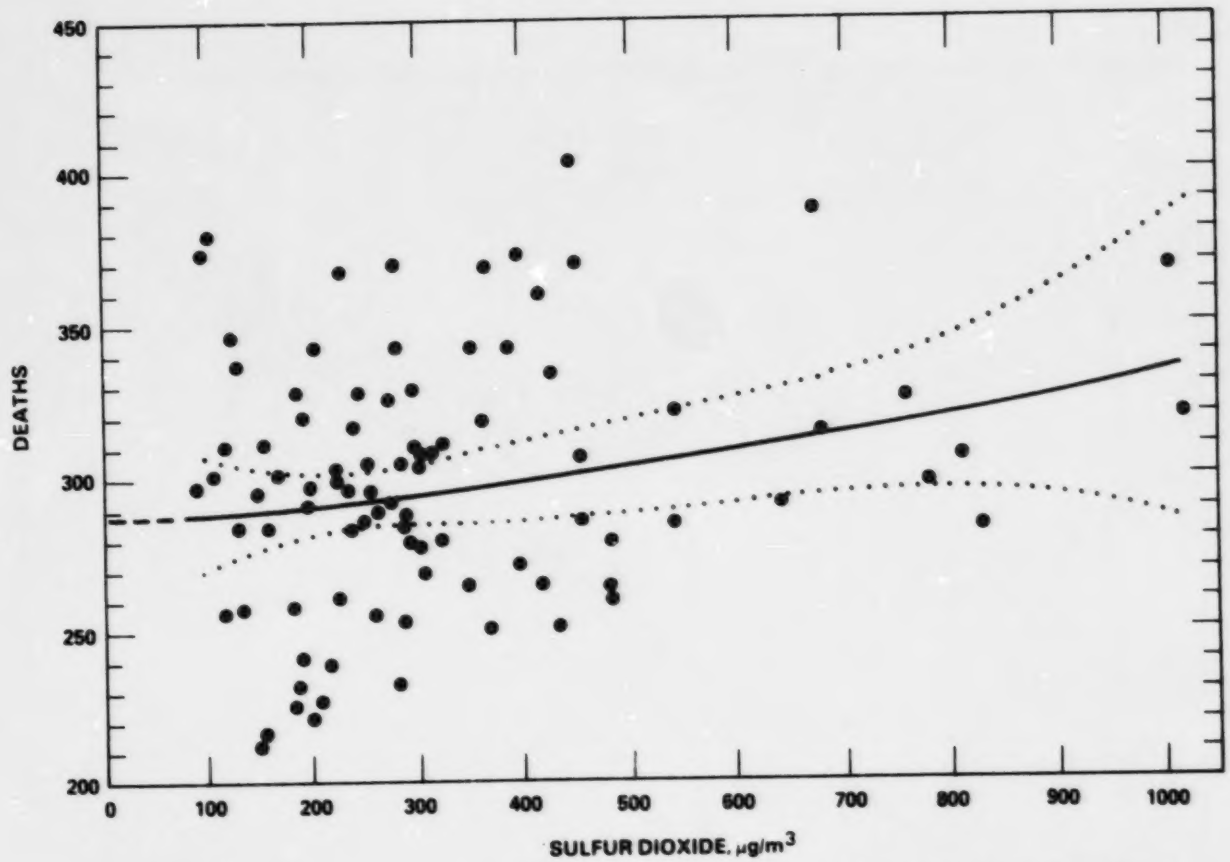


Figure 14D-4. Fitted curve and 95 percent confidence interval for Martin and Bradley data, assuming error in mortality but not in sulfur dioxide data.

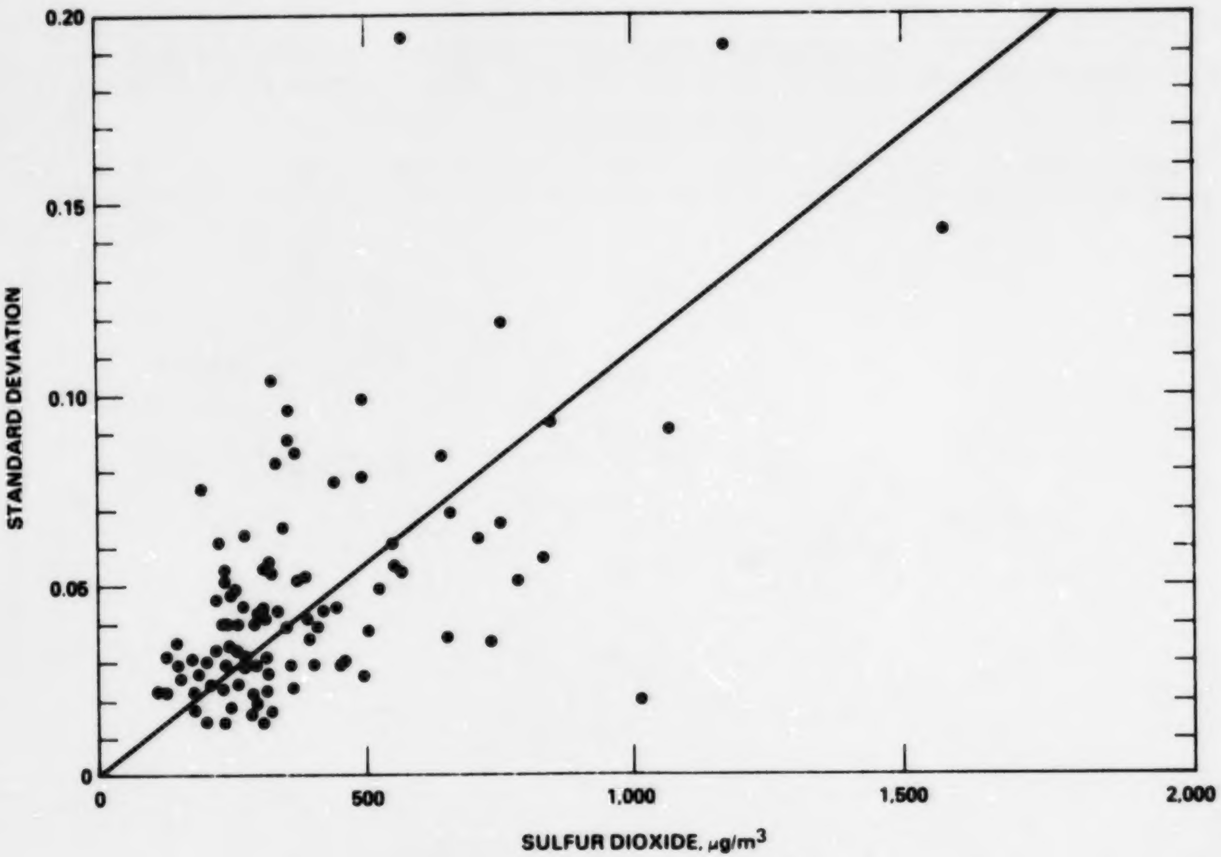


Figure 14D-5. Relationship of standard deviation to mean for SO_2 data of Martin and Bradley.

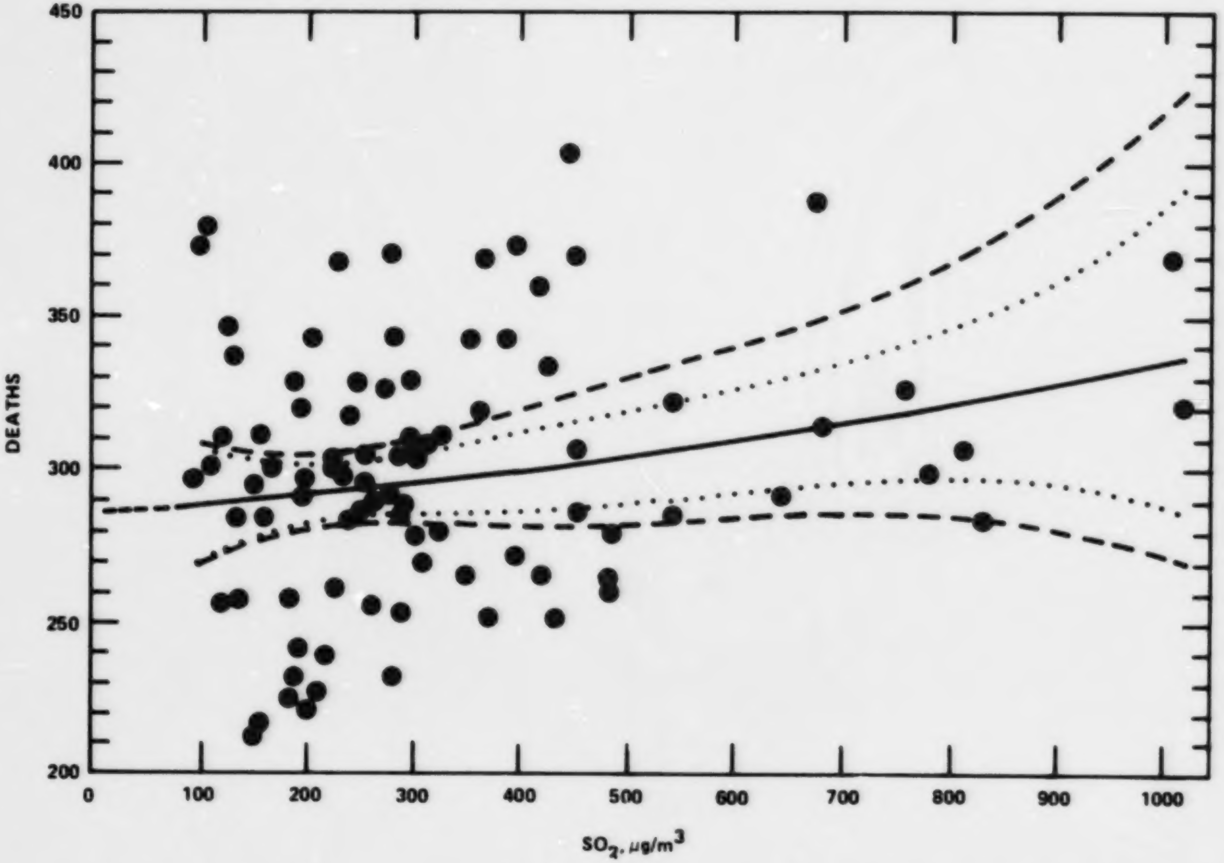


Figure 14D-6. Fitted curve (———) depicting dose-response relationship between mortality (number of daily deaths) and atmospheric sulfur dioxide concentrations (expressed in µg/m^3) during London winter of 1958/59, as determined by EPA reanalysis of Martin and Bradley (1960) data. Also shown are: 95% confidence intervals (··········) assuming error in mortality but not SO_2 data; and 95% confidence intervals (– – – –) taking into account variation or error in both mortality and SO_2 data.

APPENDIX 14E

SUMMARY OF UNPUBLISHED DAWSON AND BROWN (1981) REANALYSIS OF MARTIN AND BRADLEY (1960) DATA

A recently completed, unpublished reanalysis by Dawson and Brown of Martin and Bradley (1960) London mortality data for winter 1958-59 was presented at the November 15-17, 1981, meeting of EPA's Clean Air Scientific Advisory Committee (CASAC) as part of public participation and input concerning CASAC review of the penultimate draft version of the present criteria document. The analysis, prepared by Stanley V. Dawson, Sc.D., and Scott Brown, M.S., on behalf of the California Air Resources Board, helps to elucidate certain important points for present criteria development purposes and a summary of its key findings is included here in the Appendices of Chapter 14 for informational purposes.

The main purpose of the Dawson and Brown (1981) analysis was to reanalyze the Martin and Bradley (1960) data to evaluate possible significant relationships between London daily mortality rates and British Smokeshade (BS) measurements during a non-episodic period of four months in 1958-59, with additional emphasis on assessment of associations between mortality and 24-hour BS levels below 500 $\mu g/m^3$. Three variants of same-day models were examined: straight line, piecewise-linear, and a model log-linear for pollutants; in addition, time-series models were developed. Comparative fits of the models were principally determined by relative R^2 values, and mean squared error (MSE) was also included as a measure of error variance. The Durbin-Watson (DW) statistic was used to assess auto correlation of the error terms to determine the need for a time-series model. The mortality data were also detrended for seasonal effects, and regression analyses carried out together with determination of 95 percent confidence bands for the regression lines.

Graphs of total deaths, detrended total deaths (XTD), bronchitis deaths, detrended bronchitis deaths (XBD) and British Smokeshade (black suspended matter; BSM) levels are shown in Figure 1; note association between peaks in BSM and peaks in total death rates and bronchitis death rates. The high correlation between BSM and SO_2 levels in the data were found to preclude isolation of individual effects on mortality and, so, BSM alone was used to represent total pollution burden for a given day. Linear regression analyses were performed on detrended mortality and all levels of BSM, with the results confirming a highly significant ($P < .0001$) association between BSM and total mortality (XTD) and a smaller, but still highly significant association between BSM and bronchitis deaths (XBD). Also, the t-values were so large for the regressions XTD and XDB that a small amount of autocorrelation in errors detected by the Durbin-Watson statistic did not render the t-values insignificant. Piecewise linear models of the threshold type (hockey sticks) and of the saturation type were fitted to the data. For XTD and XBD, none of the threshold-type models gave a better fit (in terms of improved R^2-value), whereas saturation models produced improvement in fit over simple linear regressions for both XTD and XBD. Employing a logarithmic transformation of BSM, used by

Martin and Bradley (1960), and a log-linear regression model improved the fit even further over the linear model (Figure 2, 3, and 6), although one model could not be rejected for the other due to the closeness of the R^2 values for both models. Time-series models, aimed at resolving problems due to autocorrelation of the data, were found to be the best of the models considered, but coefficients of BSM and lagged BSM for such models were close to corresponding values in the linear and log-linear models without consideration of lags. Similar models were employed to evaluate the same data set, but excluding days when BSM levels were $\geq$ 500 μg/m^3. Estimates obtained were remarkably similar to those found for the whole range of pollutant levels, and models with BSM restricted to <500 μg/m^3 were still statistically significant ($P<.05$). Group means and confidence intervals are plotted in Figure 6, along with the simple linear regression line and its 95 percent confidence interval; similar means and 95 percent confidence bounds are depicted in Figure 7 for the log-linear regression plot.

The present analyses were interpreted by Dawson and Brown as being indicative of strong, statistically significant positive relationships between BSM and mortality across the whole range of BSM measurements included in the Martin and Bradley (1960) data set. Thus, although not as significant statistically due to lower signal to noise ratios at lower BSM values, there appears to be strong indications that the relationship between BSM and mortality holds for values of BSM substantially below 500 μg/m^3 and probably down to values at least as low as 200 μg/m^3. Dawson and Brown noted further that, based on the log-linear model results (providing a better fit than the simple linear model), mortality appears to increase more rapidly at lower levels of pollution for the same absolute incremental increase in BSM values. That is, an increase of 100 μg/m^3 from 200 μg/m^3 BSM appears to be more potent in causing further mortality than an increase of 100 μg/m^3 from 500 μg/m^3, and so on. It was noted that this deminishing effect of an absolute increase is common in toxicological experience, offering motivation for customary logarithmic transformtions of dose.

FIGURE 1

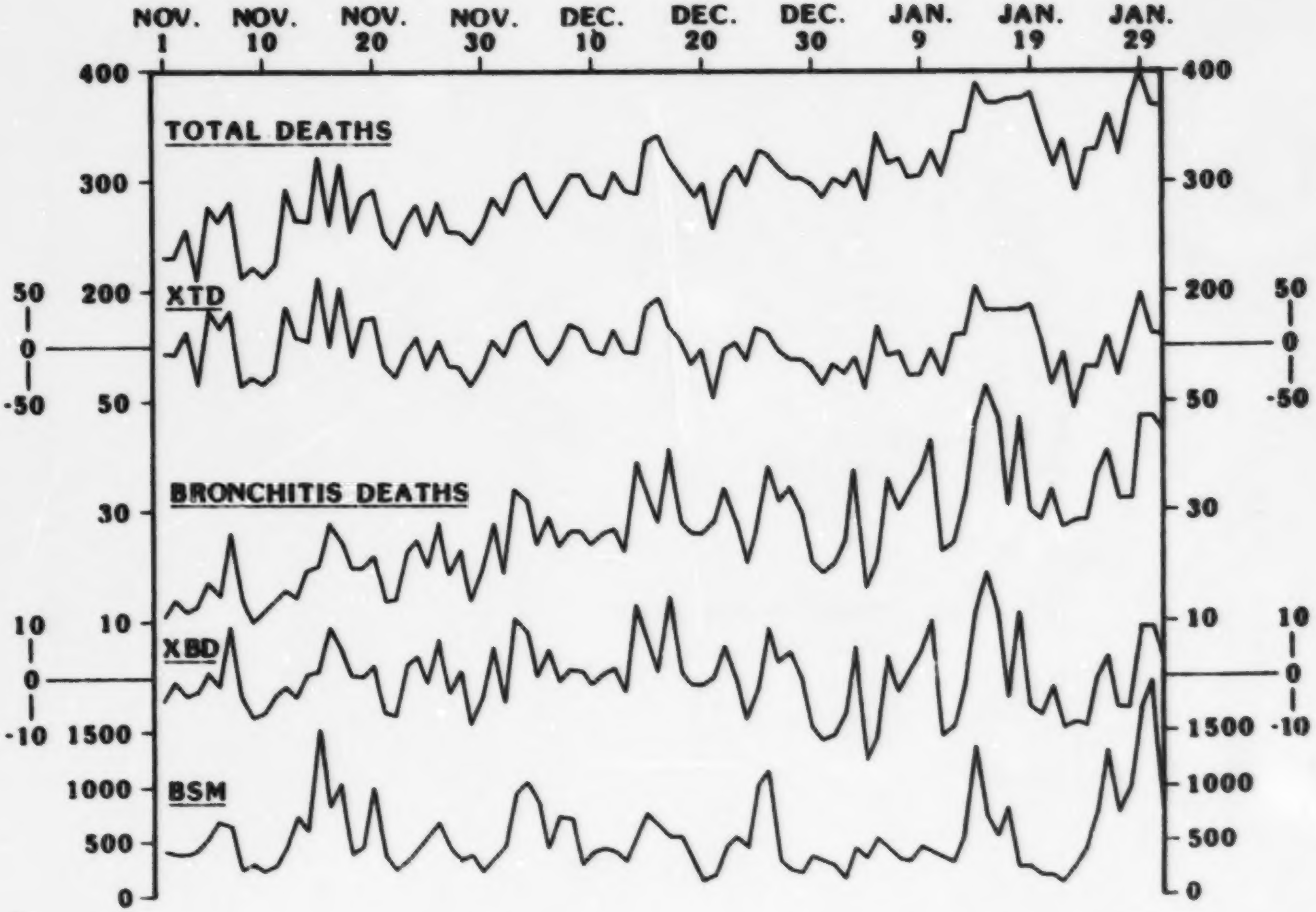

NOV. 1
NOV. 10
NOV. 20
NOV. 30
DEC. 10
DEC. 20
DEC. 30
JAN. 9
JAN. 19
JAN. 29
TOTAL DEATHS
XTD
BRONCHITIS DEATHS
XBD
BSM
400
300
200
50
30
10
1500
1000
500
0
50
0
-50
10
0
-10

FIGURE 2

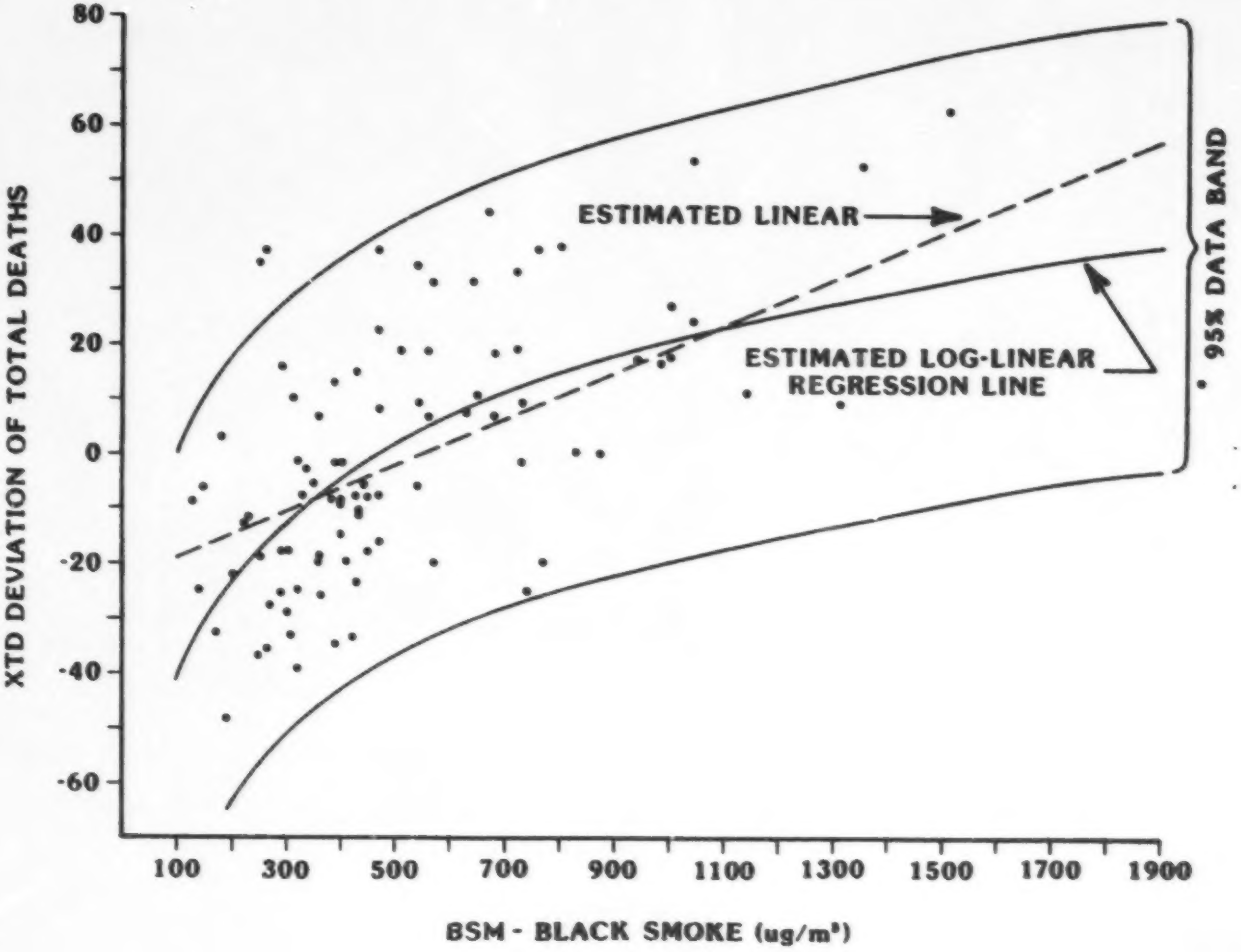

80
60
40
20
0
-20
-40
-60
XTD DEVIATION OF TOTAL DEATHS
100
300
500
700
900
1100
1300
1500
1700
1900
BSM - BLACK SMOKE (ug/m³)
ESTIMATED LINEAR
ESTIMATED LOG-LINEAR
REGRESSION LINE
95% DATA BAND

FIGURE 3

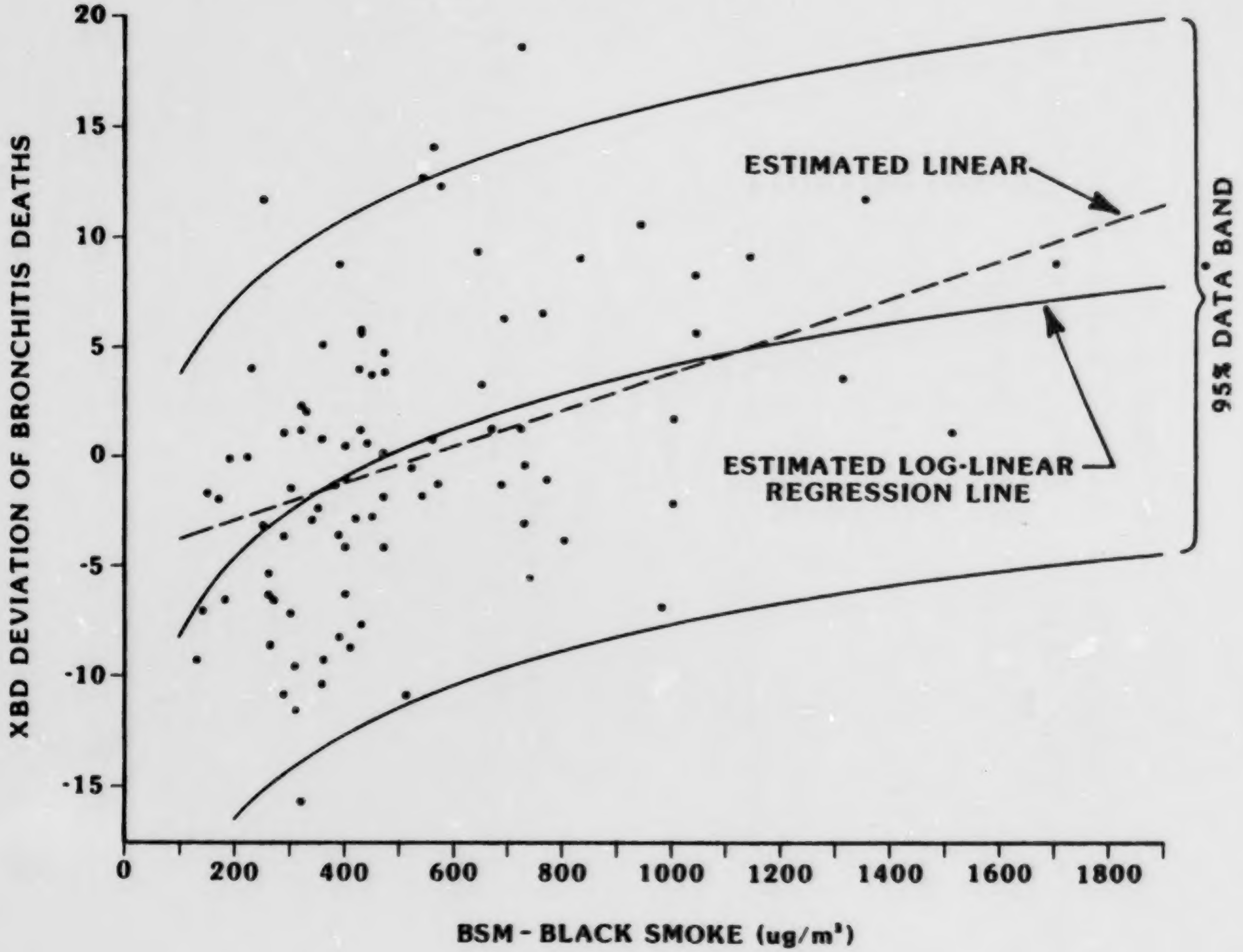

XBD DEVIATION OF BRONCHITIS DEATHS
20
15
10
5
0
-5
-10
-15
0
200
400
600
800
1000
1200
1400
1600
1800
BSM - BLACK SMOKE (ug/m³)
ESTIMATED LINEAR
ESTIMATED LOG-LINEAR REGRESSION LINE
95% DATA BAND

FIGURE 4

LEGEND: A = 1 OBS, B = 2 OBS, C = 3 OBS

Error from $XTD_t = -155.69 + .074\ XTD_{t-1} + .254\ XTD_{t-2} + 25.39 \log BSM_t$

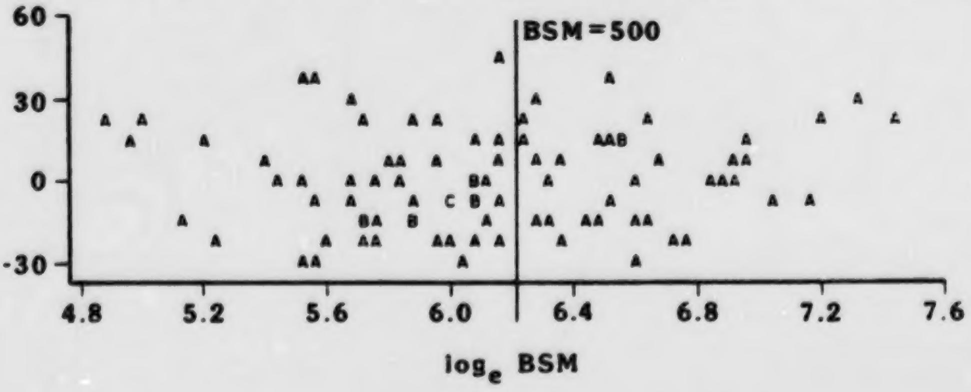


FIGURE 5

LEGEND: A = 1 OBS, B = 2 OBS, C = 3 OBS

Error from $XBD_t = -30.24 + .200\ XBD_{t-1} + 4.933 \log BSM_t$

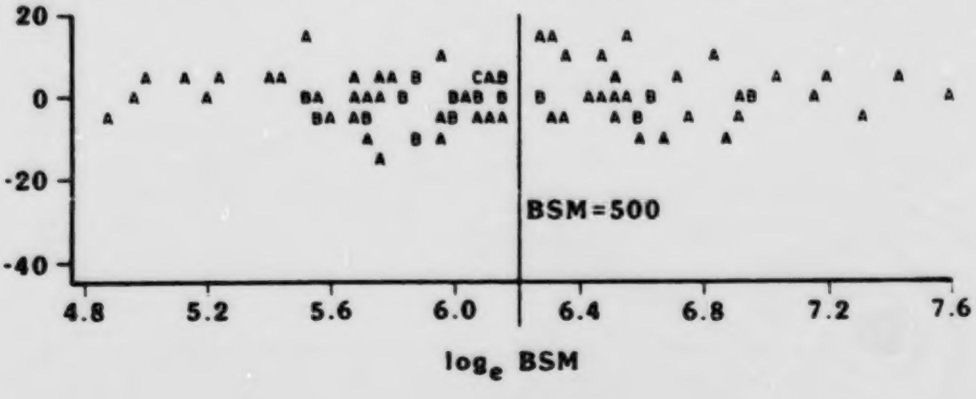


FIGURE 6

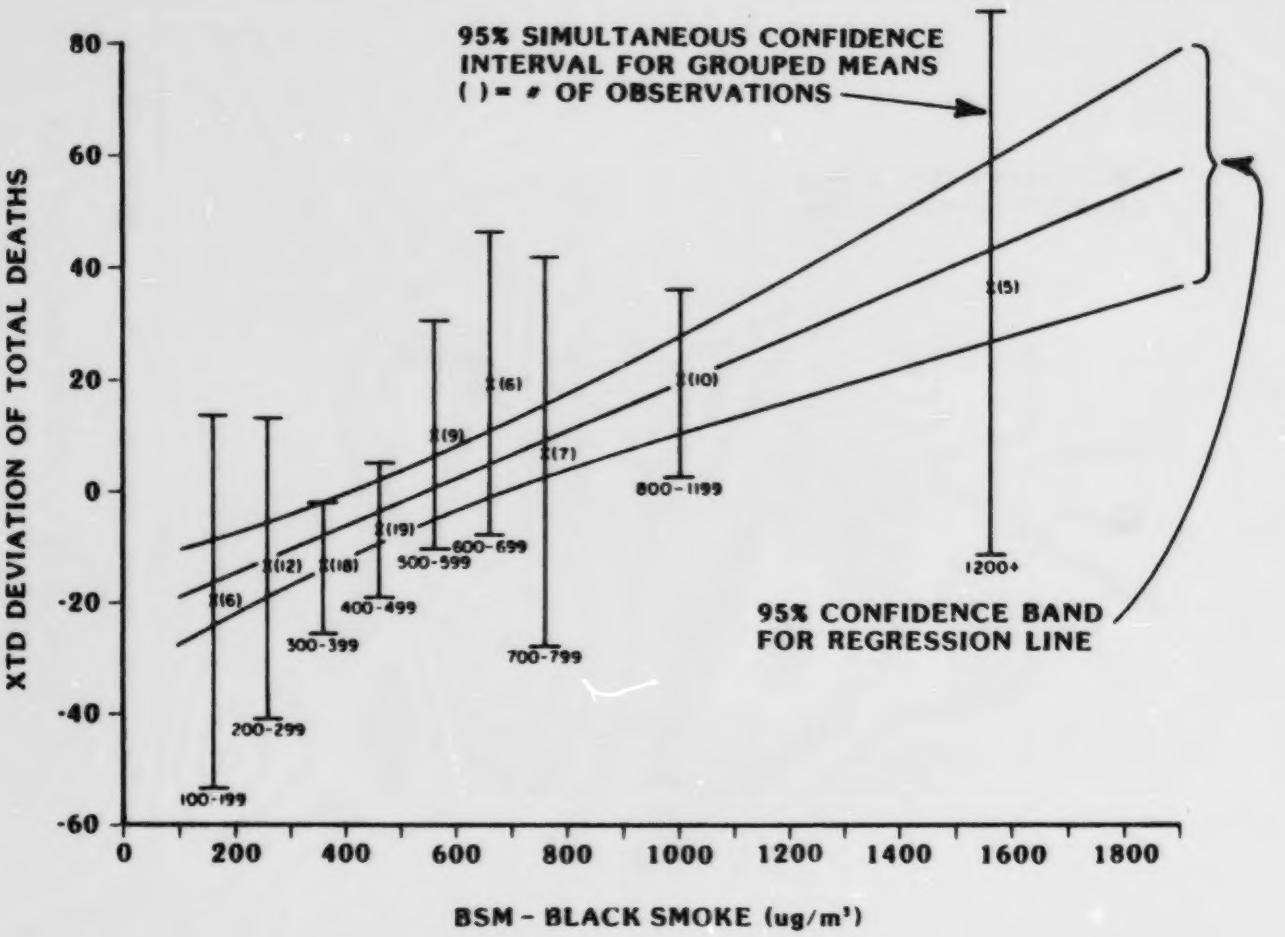
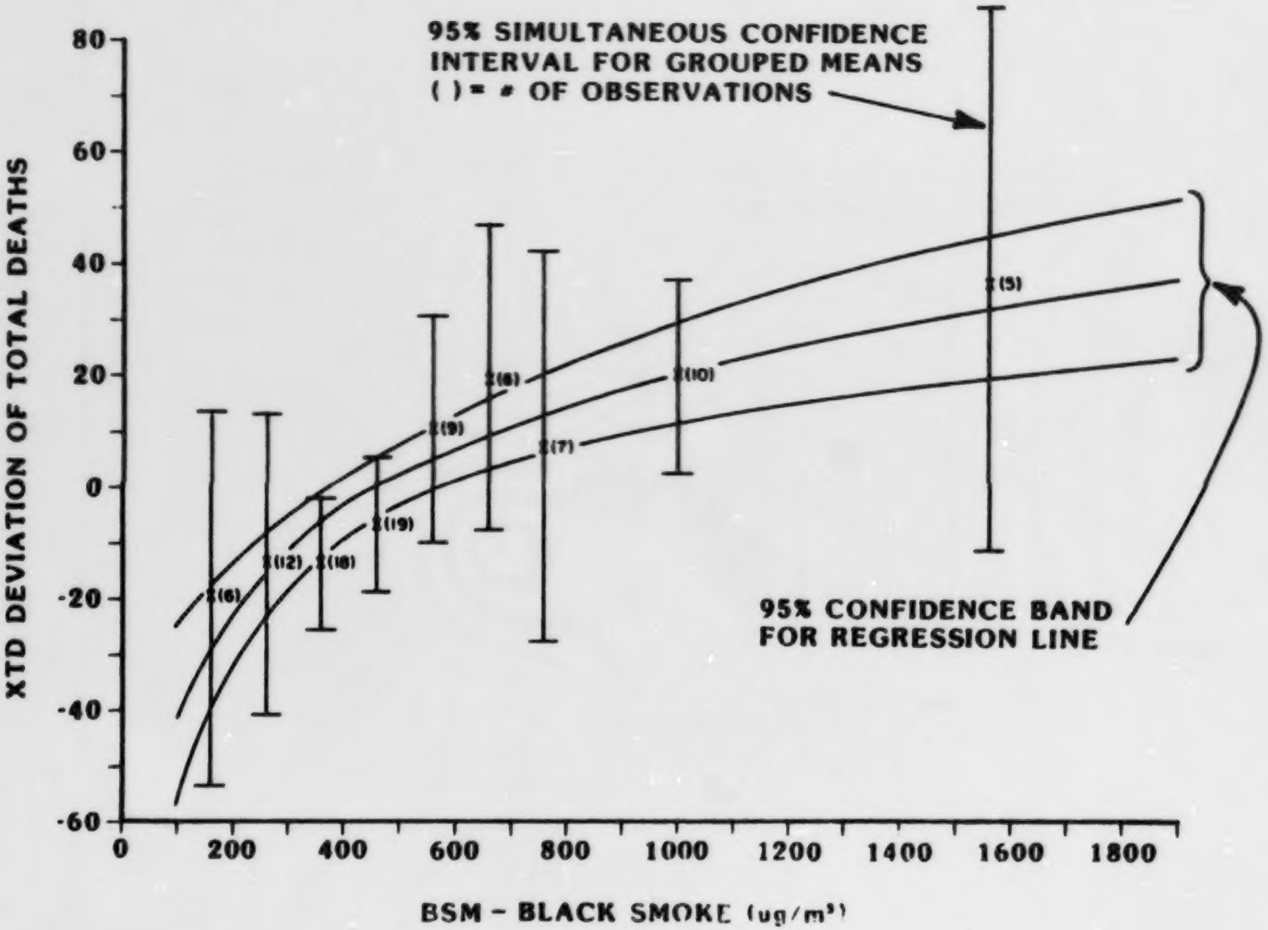

95% SIMULTANEOUS CONFIDENCE INTERVAL FOR GROUPED MEANS
( ) = # OF OBSERVATIONS
95% CONFIDENCE BAND FOR REGRESSION LINE
XTD DEVIATION OF TOTAL DEATHS
80
60
40
20
0
-20
-40
-60
0
200
400
600
800
1000
1200
1400
1600
1800
BSM - BLACK SMOKE (ug/m³)
x̄(6)
100-199
x̄(12)
200-299
x̄(18)
300-399
x̄(19)
400-499
x̄(9)
500-599
x̄(6)
600-699
x̄(7)
700-799
x̄(10)
800-1199
x̄(5)
1200+

FIGURE 7



95% SIMULTANEOUS CONFIDENCE
INTERVAL FOR GROUPED MEANS
( ) = # OF OBSERVATIONS
95% CONFIDENCE BAND
FOR REGRESSION LINE
XTD DEVIATION OF TOTAL DEATHS
80
60
40
20
0
-20
-40
-60
0
200
400
600
800
1000
1200
1400
1600
1800
BSM - BLACK SMOKE (ug/m³)
(6)
(12)
(18)
(19)
(9)
(8)
(7)
(10)
(5)

APPENDIX 14F

SUMMARY OF UNPUBLISHED ROTH ET AL. (1981) YEAR-BY-YEAR ANALYSIS OF LONDON MORTALITY DATA FOR WINTERS OF 1958-59 TO 1971-72

A recently completed, unpublished analysis by Roth (1981) of London mortality data in relation to British Smokeshade (BS) and SO_2 levels during the winters of 1958-59 to 1971-72 (using the same data set evaluated by Mazumdar et al., 1981) was presented at the November 15-17, 1981, meeting of EPA's Clean Air Scientific Advisory Committee (CASAC) as part of public participation and input concerning CASAC review of the penultimate draft of the present criteria document. The analysis, prepared by H. Daniel Roth, Ph.D. (of Roth Associates, Inc.) on behalf of the Utilities Air Regulatory Group (UARG), helps to elucidate certain important points for present criteria development purposes and a summary of key findings of the Roth analysis is included here in the Appendices of Chapter 14 for informational purposes.

The main purpose of the Roth analysis was to evaluate, on a year-by-year basis, relationships between daily mortality and BS or SO_2 levels occurring in London during the winters of 1958-59 to 1971-72. Multiple regression analyses were employed, taking into account temperature, humidity, high correlations between BS and SO_2 levels, and certain temporal trends in the data. The results of the analysis are concisely summarized in Table 14F-1. Statistically significant ($P < .05$) associations were found between mortality derivations and either temperature or humidity or both for several of the years evaluated by Roth. Based on this, Roth indicated that reanalyses (discussed earlier in Chapter 14 or Appendix 14D) of Martin and Bradley (1960) London mortality data by Ware et al. (1980) or by Hasselblad and Stead (see Appendix 14D) cannot be accepted as demonstrating quantitative exposure-response relationships between deviations in London 1958-59 winter mortality rates and concurrent BS or SO_2 levels, because the two reanalyses did not take into account the effects of temperature or humidity (shown by Roth to be potentially important confounding variables) or other potentially important factors (e.g. lag effects).

It should also be noted, however, that Roth's results also equally suggest that significant associations may exist between mortality deviations in London and concurrent levels of BS or SO_2 during various winters from 1958-59 to 1971-72. For example, significant positive associations between mortality and SO_2 levels were found by Roth for the 1958-59 and 1962-63 winters, suggesting that positive mortality-SO_2 associations detected in analyses of the Martin and Bradley (1960) data set for the 1958-59 winter occurred, although they may be somewhat unusual in comparison to other London winters. Roth's analysis also suggests that statistically significant positive associations may exist between daily mortality and BS levels for several additional winters, including the 1967-68 winter when 24-hour BS levels rarely (if ever) exceeded 500 $\mu g/m^3$. EPA interprets the latter finding as tending to confirm and reinforce indications based on the analyses of Mazumdar et al. (1981) and others discussed in Chapter 14 that positive associations exist between mortality and London BS values below 500 $\mu g/m^3$ (and possibly as low as 150-200 $\mu g/m^3$), although none of the different individual analyses can be said to conclusively delineate precise dose-response or exposure-response relationships between London mortality and BS or SO_2 levels.

TABLE 14F-1
MULTIPLE REGRESSION ANALYSIS OF DEVIATIONS OF MORTALITY VERSUS DEVIATIONS OF AIR QUALITY VARIABLES

TIME PERIOD	STATISTICAL SIGNIFICANCE OF ASSOCIATION (SIGN OF ASSOCIATION)			
	TEMPERATURE	RELATIVE HUMIDITY	SO_2	BRITISH SMOKE
Winter 1958-59	S*		S	
Winter 1959-60			-	
Winter 1960-61			-	S
Winter 1961-62	S			
Winter 1962-63	S	S	S	-
Winter 1963-64	S	-**	-	S
Winter 1964-65		-	-	S
Winter 1965-66	S	S		
Winter 1966-67	S			
Winter 1967-68		S	-	S
Winter 1968-69	S			
Winter 1969-70	S	-		-
Winter 1970-71		S		
Winter 1971-72	S			

*Significance at the .05 level
**Negative association

APPENDIX 14G

SUMMARY OF MAZUMDAR ET AL. YEAR-BY-YEAR ANALYSIS OF LONDON MORTALITY DATA FOR WINTERS OF 1958-59 TO 1971-72

A third analysis of London mortality data in relation to PM and SO_2 levels (in addition to the two analyses summarized in Appendices 14E and 14F) was presented at the November 15-17, 1981, meeting of EPA's Clean Air Scientific Advisory Committee (CASAC) as part of public participation and input concerning CASAC review of the penultimate draft of the present criteria document. That third analysis, summarized here, was presented by Dr. Ian Higgins at the CASAC meeting in order to provide an update regarding newly emerging results derived from the work of Mazumdar et al. completed since the publication of the Mazumdar et al. (1981) report alluded to in the main text of Chapter 14.

Dr. Higgins described the results of a year-by-year analysis of the same 1958-59 to 1971-72 London winter mortality data previously analyzed by Mazumdar et al. (1981) but not on a year-by-year basis. Certain allowances were incorporated into the analyses in order to control for potentially confounding factors, e.g.: year to year variation in background mortality rates; seasonal trends; temperature and humidity effects; lag effects; and day of week or month of winter effects. Compared against 7-year averages of yearly coefficients for % mortality per mg/m^3 of each pollutant, notable increases (10-70%) in mortality over the 7-year means were observed in relation to BS levels for most of the winters between 1958-59 and 1971-72, but smaller increases (10-20%) in mortality occurred in relation to SO_2 levels only during about ½ of the winters. The results of regression analyses evaluating a change in % mortality as a function of varying BS and SO_2 levels over the entire 14 winters analyzed are summarized in Figures 14G-1 and 14G-2, which respectively show (1) variations in daily mortality rates in relation to variations in particulate matter (BS) levels, holding concurrent SO_2 levels constant and (2) variations in daily mortality rates in relation to SO_2 levels, holding concurrent BS levels constant.

Dr. Higgins noted that the increase in mortality with increasing smoke concentrations over different SO_2 concentrations is clear (as seen in Figure 14G-1). He further noted that (as seen in Figure 14G-2) between about 600 and 800 $\mu g/m^3$ (0.6 and 0.8 mg/m^3) SO_2 mortality is flat for concentrations of smoke of 500 $\mu g/m^3$ (0.5 mg/m^3) and above 500 $\mu g/m^3$ mortality tends to increase with increasing SO_2 concentrations (possibly due to interaction between smoke and SO_2). Taking this information into account and the results of the earlier regression analyses reported by Mazumdar et al. (1981), as shown in Figures 14-2 and 14-3 of Chapter 14 of the present document, Dr. Higgins concluded that: (1) the quadratic model dose/response curve of Mazumdar et al. (1981) shows a possible threshold at 300 $\mu g/m^3$ BS (though any material increase in mortality appears to occur only at much higher levels); and (2) at daily pollutant (BS and SO_2) concentrations of less than 500 $\mu g/m^3$, any conclusions about dose-response relationships for mortality should be drawn with caution.

Figure 14G-1.

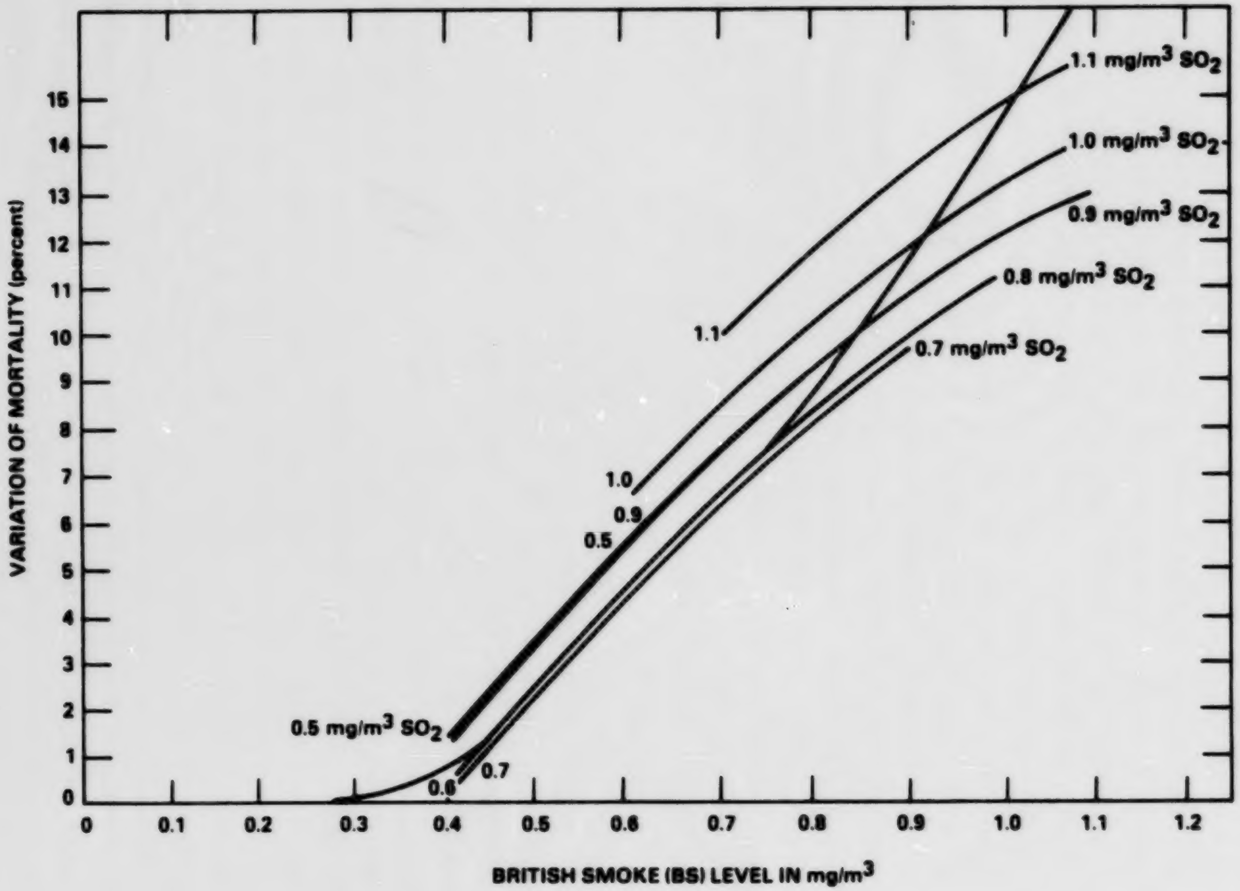
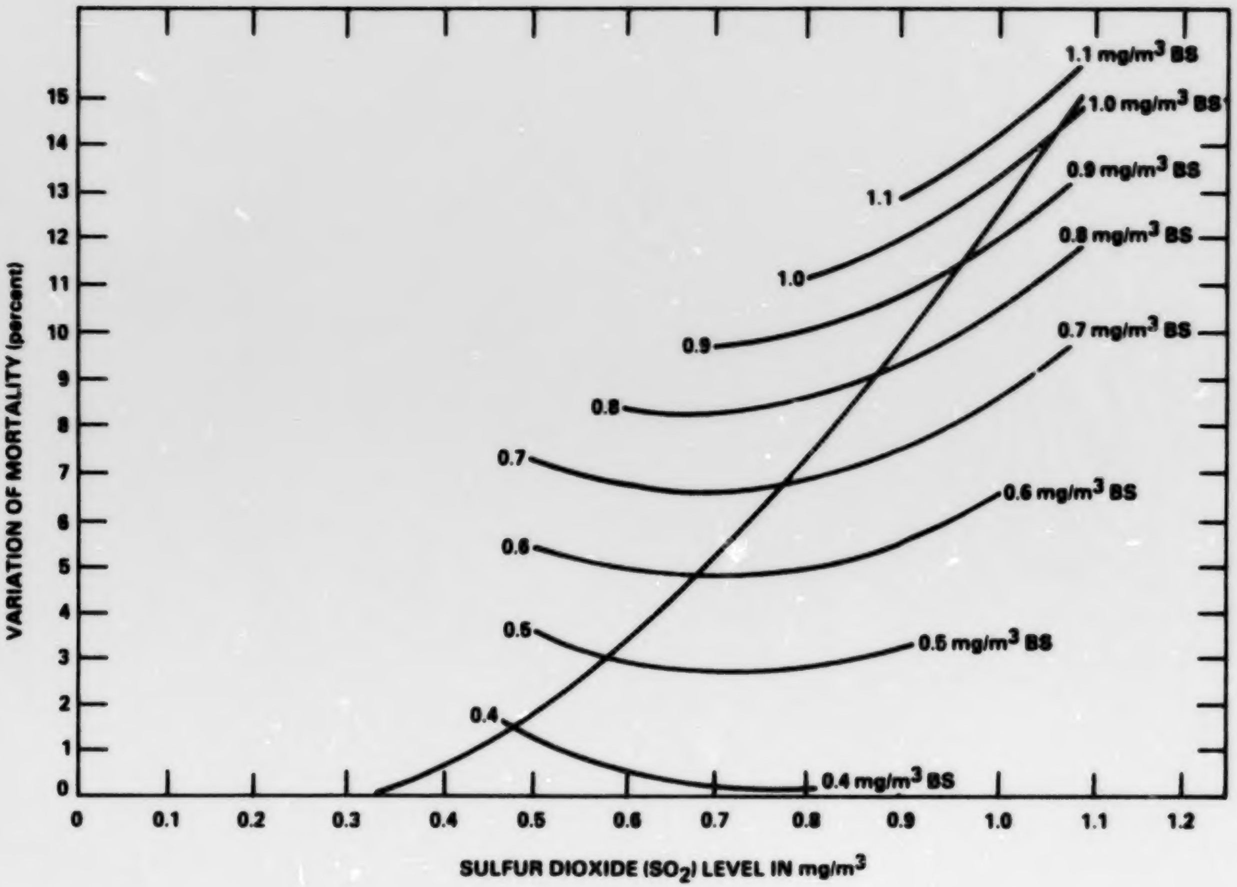

VARIATION OF MORTALITY (percent)
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
BRITISH SMOKE (BS) LEVEL IN mg/m3
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
1.1
1.2
1.1 mg/m3 SO2
1.0 mg/m3 SO2
0.9 mg/m3 SO2
0.8 mg/m3 SO2
0.7 mg/m3 SO2
1.1
1.0
0.9
0.5
0.5 mg/m3 SO2
0.6
0.7

Figure 14G-2.


VARIATION OF MORTALITY (percent)
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
SULFUR DIOXIDE (SO_2) LEVEL IN mg/m^3
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
1.1
1.2
1.1 mg/m^3 BS
1.0 mg/m^3 BS
0.9 mg/m^3 BS
0.8 mg/m^3 BS
0.7 mg/m^3 BS
0.6 mg/m^3 BS
0.5 mg/m^3 BS
0.4 mg/m^3 BS
1.1
1.0
0.9
0.8
0.7
0.6
0.5
0.4